A review of fisheries enhancement methods to promote profitability and sustainability in Australian fisheries

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This publication has been compiled by Andrew Norris and Michael Hutchison of Agri-Science Queensland, Department of Agriculture and Fisheries.

The Department of Agriculture and Fisheries proudly acknowledges all First Nations peoples (Aboriginal peoples and Torres Strait Islanders) and the Traditional Owners and Custodians of the country on which we live and work. We acknowledge their continuing connection to land, waters and culture and commit to ongoing reconciliation. We pay our respect to their Elders past, present and emerging.

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Abbreviations and definitions

Altruism value	The preference of the individual for others of the current generation to enjoy and benefit from a resource, even if the individual professing the value does not use the resource themselves.
Benefit cost ratio (BCR)	The ratio of net present benefits to net present costs.
Benefit transfer	Use of benefit estimates from one or more studies at sites where primary research was conducted to estimate benefits at a different site of interest.
Bequest value	The preference of the individual for others of future generations to enjoy and benefit from a resource, even if the individual professing the value does not use the resource themselves.
Consumer surplus	Enjoyment experienced from the outcomes of the economic activity and is a measure of the value of those goods or activities to the end user.
Contingent valuation (CV)	A stated preference technique where people are directly asked their willingness to pay or accept compensation for some change in an ecosystem service.
Cost effectiveness analysis (CEA)	Cost effectiveness analysis assesses the impact of different options in physical terms, and compare these to the costs of the different options to determine which option, or mix of options, achieves the target at least cost.
Cultural services	Typically non-material benefits received by people from direct and indirect interactions with wetlands such as recreation, aesthetic values, spiritual benefits and enhancements in knowledge
Direct use values	Direct use values measure the willingness to pay for the good as a final consumption good.
Discrete Choice Experiments (DCE)	A quantitative technique for eliciting preferences involves asking individuals to state their preference over hypothetical alternative scenarios, goods or services.
Ecosystem services	Ecosystem goods (such as food) and services (such as waste assimilation) that represent the benefits human populations derive, directly or indirectly, from ecosystem functions.
Employment	The change in the number of jobs generated in a region resulting from a change in regional output, expressed on an annual full time equivalent basis.
Existence value	Existence value refers to the willingness to pay to keep a good in existence in the context where the individual expressing the value has no actual or planned use of the resource for herself, or for anyone else.
Indirect use values	Indirect use value measures the value that a good has as an intermediate input in some production process whose end good is of value.
Labour income	Employee wages and salaries, including payroll benefits, and income of sole proprietors.
Market value (MV)	Values are directly obtained from what people must be willing to pay for the service in a market transaction.
Non-use value	Refers to the willingness to pay to maintain some good in existence even when the individual does not use the resource or plan to use the resource at some time in the future. Non-use values are generally separated into existence, altruism and bequest values.

Net present value (NPV)	An economic term representing the total economic value of an item over time (benefits - costs), discounted to present day terms.
NRM	Natural Resource Management.
Option value	Option value relates to retaining an option for that resource use in the future.
Output	The value of industry production. It is the sum of all intermediate sales (business to business) and final demand (sales to consumers and exports).
Producer surplus	Sometimes called economic rent. Equals the total revenue minus total costs for commercial operators (Milon 1991), and is therefore the change in net income or profits following an activity.
Production function	The production function approach values ecosystem services as inputs into another production process, and focuses on estimating those ecosystem services arising from the regulatory and habitat functions of ecosystems.
Regulating services	Essentially the benefits to humans attributable to the regulation of ecosystem processes such as water treatment and local climate regulation.
Revealed preference (RV)	Economic techniques that are based on the assumption that the preferences of consumers can be revealed by their purchasing habits *e.g.* Travel cost.
Stated preference (SP)	Economic techniques that elicit consumer preferences through surveys in which respondents state their preferences in response to hypothetical scenarios, e.g Contingent valuation.
Supporting services	Services that underpin the production of all other ecosystem services such as nutrient cycling, water cycling, and provisioning of habitat.
Total Economic Value (TEV)	An economic framework that identifies not only the value of financial or commercial outputs, but also non-consumptive values that may be environmental or social in nature.
Travel cost (TC)	Values of site-based amenities are implied by the costs people incur to enjoy them. A recreation area can be valued at least by what visitors are willing to pay to travel to it, including the imputed value of their time.
Use values	Use values measure the value arising from the actual, planned or possible use of goods and services. Use values can be direct, indirect, or option values.
Value-added	The difference between the amount an industry sells a product for and the production cost of the product. Value added measures contribution to Gross Domestic Product (GDP), and is the preferred measure of economic impacts on a regional economy because it includes all sources of income to the region.
Wetland	Areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine waters. Wetlands account for a wide variety of habitat types including rivers, shallow coastal waters and coral reefs.
Willingness to pay (WTP)	The price or dollar amount that someone is willing to give up or pay to acquire a good or service.

Executive summary

Background

Increasing stress is being placed on the profitability and long-term sustainability of many Australian fisheries. Fish stocks and therefore fisheries productivity can be constrained by factors such as recruitment, habitat, trophic webs and genetic bottlenecks. Even well-managed fishery stocks are unlikely to yield increased harvests in the immediate future using traditional management approaches. Increasing attention is therefore being directed towards pro-active fishery management options.

Fisheries enhancement refers to the deliberate application of measures aimed at enhancing productivity and long-term sustainability beyond what is achievable by good harvest management alone. Fisheries enhancement strategies expand the options available to fisheries managers beyond the use of traditional input-output controls. They provide opportunities for significant socio-economic benefits, through actively improving aquatic habitat and management of fish at the population level. Such approaches may simply offer alternative routes to a particular outcome, or they may support or create outcomes that cannot be achieved by other fisheries management measures (*e.g.* stocked impoundment fisheries). Enhancement strategies also have the potential to help manage the sometimes high social costs associated with harvest regulations.

Within Australia, fishery enhancement strategies have been applied across a variety of fisheries, but broad and consistent uptake has been limited. Constraints to uptake include fishery manager knowledge levels and their ability to incorporate information on relative merits of different enhancement techniques into their fisheries management decision making processes. Quantitative comparison will enable decisions to be made with greater certainty and to deliver the best value from an investment.

Objectives

The objective of this review was to provide a consolidated fisheries enhancement knowledge base to enable robust comparisons of the relative return on investment for different enhancement approaches across various fisheries, and assessment of their long-term viability and impact on fishery productivity and sustainability. Increasingly, it is recognized that fisheries should be managed for socio-economic objectives in addition to biological objectives. The focus of this review was therefore to synthesize information on the socio-economic costs and benefits of fisheries enhancement strategies to provide managers with a more comprehensive suite of data to inform their management decisions.

Methodology

A systematic review of literature was undertaken to identify quantitative data on the costs, benefits and socio-economic evaluations of fisheries enhancement projects both within Australia and globally. The review targeted three categories of enhancement activities: habitat enhancement with artificial reefs and fish attracting devices (FADs), fish stocking, and rehabilitation of natural habitat. Within each of these categories, data was collated from both academic (peer-reviewed) and professional (technical) literature. Economic valuation and economic impact analysis were used to compare the economic outcomes to fisheries between projects. Whilst a broad range of ecological and socio-economic benefits can be generated by enhancement projects, this review focussed only on the benefits accruing to fisheries. As long as the net benefits to fisheries exceeded the net costs of implementation, the enhancement activity will provide positive net benefits from a fisheries management perspective.

In total, 224 articles were identified that quantitatively reported on socio-economic parameters of fisheries enhancement projects. Where sufficient information was available, data within each category was further grouped to provide an overview of socio-economic benefits for various actions or scenarios. For each group, a cost benefit analysis was conducted using mean or median values, based on a 30 year time horizon and 5% discount rate. The results from the cost benefit analyses were used to compare the indicative NPV and BCR between groups and techniques, to help inform managers of the relative potential for each approach.

Key findings

A measured and responsible approach to the employment of fisheries enhancement strategies has generally been undertaken in Australia, particularly in the past two decades. Comprehensive research and planning now underpins most new enhancement projects (e.g. NSW marine stocking strategy, WA artificial reef program etc.). However, better data collection on the socio-economic impacts is required to facilitate greater uptake by fishery managers and develop support from stakeholders. New projects need to incorporate socio-economic appraisal or evaluation as a core component of their design, not only to further our knowledge base, but to also justify to stakeholders and investors that the expenses outlaid have been warranted and will provide a positive socio-economic return.

Habitat enhancement has primarily been undertaken for recreational fisheries in Australia. The installation of artificial reefs typically increase fish abundance, biomass and diversity when installed at sites where the existing habitat is bare or homogenous. Recent research clearly demonstrates that purpose-built reefs, constructed of sufficient size and complexity, are capable of both attracting and producing fish of recreational and commercial importance. There is a growing consensus that most artificial reef projects have warranted the expense. However, the economic value of habitat enhancement projects whilst positive, typically returns relatively low benefit cost ratios (media BCR = 1.29, internal rate of return = 8.55%).

Stocking practices in Australia can generally be considered to be comply with world best practice, but require better socio-economic evaluation. Strategically planned fish stocking can make significant contributions to fishery catches and deliver substantial socio-economic benefits under the right circumstances. However, there are significant ecological risks that need to be well-managed, especially genetic impacts on wild population. The majority of fish stocking in Australia has occurred for freshwater recreational fisheries, but estuarine stocking is becoming more prevalent. Our understanding of the full impacts of stocking has been hindered by the inability to discriminate between hatchery-reared and wild fish. Angler willingness to pay has demonstrated the socio-economic feasibility and public support of stock enhancement programs for recreational fisheries improvement. Stocking produced the highest benefit cost ratios (median BCR = 7.41). The greatest economic benefits were achieved for recreational fisheries in enclosed waterbodies (*e.g.* freshwater impoundments) with limited natural recruitment and where emigration of stocked fish was restricted. Few studies have demonstrated stocking to have an additive effect on regional fish abundance. Three key components should be incorporated to maximise the cost-efficiency and potential socio-economic feasibility of future fish stocking programs: 1) pilot studies to inform optimal stocking strategies, 2) bio-economic modelling to maximise economic feasibility, and 3) application of an adaptive management framework to capitalise on new opportunities, address unforeseen threats and variations in natural recruitment.

Large-scale habitat enhancement conducted from a fisheries perspective has experienced slow uptake In Australia, in part due to manager uncertainty on the likely outcomes. The socio-economic costs and benefits are rarely fully quantified, making it difficult to compare justification of costs between different projects or management options. Mangrove rehabilitation is likely to generate the best return on investment from a fisheries perspective. Rehabilitation of rivers habitat, seagrass and shellfish reefs are also likely to provide positive economic returns for fisheries, but the high cost of coral reef restoration and the low value of fishery production from salt marshes mean rehabilitation of these habitats is not likely to be economically feasible for fisheries enhancement. These results are only indicative, and care needs to be taken because they are sensitive to the input values of the cost and benefits. The benefit-cost ratio for preserving natural habitats can be as high or higher than rehabilitation activities, suggesting that investing in maintaining and preserving existing natural habitats may in some instances be more cost-effective and deliver greater benefits than rehabilitating degraded systems. There is considerable community support for utilising habitat rehabilitation to enhance fisheries, with a willingness to pay amongst both users and non-users. Such user or community support is vital to encourage backing by politicians and uptake of habitat rehabilitation as a fisheries enhancement tool by managers.

Clearly understanding the threats and stressors impacting fisheries systems, identifying realistic and quantitative management objectives and increased use of bio-economic modelling will be core to pro-actively managing commercial and recreational fisheries in Australia in a sustainable way using fishery enhancement strategies. Incorporating enhancement options into decision making processes will expand the options available to fisheries managers beyond the use of traditional input-output controls.

There is potential to expand the value of recreational fisheries and create niche fishing opportunities that can drive regional development. Rehabilitation of aquatic habitat has the potential to sustainably increase the productivity, yield and value of Australia’s fisheries and improve resilience against adverse events and climate change. Fish stocking has significant potential for expansion, particularly in closed and semi-closed estuarine and impoundment systems. There are opportunities to expand the suite of species that are currently stocked, to diversify recreational fishing opportunities and attract more angler to areas with unique fisheries. Artificial reefs and FADs currently have high recreational fisher support, and their installation can improve access and fishing options. Undertaken appropriately, habitat enhancement can also increase overall fishery productivity by supporting the life-history requirements of fish and invertebrates.

For commercial fisheries, habitat rehabilitation has the potential to increase wild recruitment that may be limited due to the degradation and loss of essential fish habitats, such as nursery areas. Better recruitment is likely to result in increased yields and greater long-term sustainability within fisheries. Stock enhancement has the potential in some species to help recover depleted wild stocks more rapidly or to create new fisheries. The greatest potential for stock enhancement in the short-term remains with stocking less mobile, high-value invertebrate species.

In this review enhancement activities have been viewed purely through the lens of the resultant socio-economic benefits to fisheries. However, enhancement activities can deliver a broad range of ecosystem service benefits which provide substantial value beyond fisheries. These activities can also improve environmental health, species conservation, and support provision of a suite of other environmental, social, recreational and commercial opportunities. The cost benefit results in this review have deliberately not taken these additional benefits into account, in order to provide a clearer picture of the outcomes for fisheries. However, the substantial additional or value added benefits that can be generated can be used to seek and justify co-investment in projects from non-fishery sectors. Collaborating with relevant non-fishery stakeholders has the potential to greatly reduce the direct contribution costs of fisheries managers for some enhancement projects, which would lead to significantly better benefit cost ratios than those reported in this review. Where possible fisheries enhancement projects should pro-actively seek support from non-fishery sectors to more cost-efficiently achieve their fishery management objectives.

Fishery enhancement approaches should not be seen as a replacement for good fishery management, but instead as part of a suite of potential management tools that can be utilised together to deliver strong, sustainable fisheries outcomes. Integrating different fishery enhancement strategies has the potential to deliver substantial socio-economic benefits. The greatest benefits are likely to occur when different strategies are integrated to comprehensively address the issues limiting fisheries production or expansion. Combining management of habitat to increase carrying capacity and responsible stock enhancement to overcome recruitment limitations will help optimise stock levels and harvest potential in the most efficient way. Bio-economic modelling is key to appraising the potential of integrated habitat or stocking initiatives relative to other fisheries management measures and evaluating the cost–benefits of individual programs.

Recommendations

The following recommendation will help clarify our understanding on the outcomes of fisheries enhancement activities and enable more cost-effective implementation:

- All new major projects should incorporate some form of socio-economic analysis to understand the outcomes of the activities, and develop a knowledge database that can assist in the feasibility analysis of new projects.
- Implementation of stocking cost-efficiency gains should be undertaken by improving release strategies through better survival and fitness from pre-release training, acclimation and improvement of release habitats.
- Standardised analysis and reporting guidelines should be developed to provide consistent and comparable results.
- Adaptive management should be employed in fishery enhancement projects to enable the results from monitoring and research to be rapidly adopted to maximise outcomes.
- A suitability matrix for fisheries enhancement options should be developed for the various fisheries in Australia to provide managers with a rapid method for identifying appropriate enhancement strategies.
- The potential of developing a generic bio-economic model and an associated database containing relevant biological and economic parameters for a range of species and fisheries should be investigated.

- Co-investment for fisheries enhancement projects should be sought from various stakeholder groups where relevant, to capitalise on the broad ecosystem services that can be delivered.
- The use of different fisheries enhancement strategies should be integrated to potentially deliver multiplicative benefits across the entire life-history of target species. Habitat and recruitment are both essential to achieve sustained fishery outcomes, and enhancement projects should integrate both where possible to improve fitness across the entire life-cycle of target species.
- A national approach to identification of hatchery-reared fish is critically needed. Genetic marking through lineage and parental analysis holds great promise and should be considered for national adoption.

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Chapter 1. Introduction

Many fisheries resources worldwide are fully or over-exploited, and Australia is no exception (FAO 2022). Fisheries resources are experiencing ever-growing pressure from population growth, habitat degradation, access restrictions, climate change, rising consumer demand, expanding focus on food security, economic development and overfishing (Munro and Bell 1997, Crowder *et al.* 2000, Morgan *et al.* 2001, Brown and Day 2002, Post *et al.* 2002, Molony and Bird 2002, Molony *et al.* 2003, Bell *et al.* 2006, Lorenzen *et al.* 2013). These are placing increasing stress on the profitability and long-term sustainability of many fisheries. Fish stocks and therefore fisheries productivity can also be constrained by factors such as recruitment, habitat, trophic webs and genetic bottlenecks (Becker *et al.* 2018). Even well-managed fishery stocks are unlikely to yield increased harvests in the immediate future. It is likely that future demands on fisheries resources will intensify public pressure to augment natural resources through enhancement activities. Increasing attention is therefore being directed towards pro-active management options to potentially alleviate these constraints.

Traditional input-output management approaches will continue to play an important role in helping to sustain fisheries in the face of these pressures, but it is unlikely to have the capacity to fully mitigate the broad challenges faced (Hollowed *et al.* 2013). Over the last few decades there has been an increasing shift towards the utilisation of techniques that involve manipulation of aquatic environment and direct enhancement of fishery stocks (Ross 1997, Lorenzen 2014).

A range of techniques have been employed around the world to enhance the value and sustainability of recreational and commercial fisheries (Florisson *et al.* 2018). In many instances these techniques have proven to be extremely effective and become core components of fisheries management (Florisson *et al.* 2018). In other instances, successful results have not been achieved or the risks have been considered to outweigh the potential benefits.

Fisheries enhancement refers to the deliberate application of measures aimed at enhancing productivity and long-term sustainability beyond what is achievable by good harvest management alone (Taylor *et al.* 2018). In theory, successful fisheries enhancement has the potential to yield significant productivity, social and ecological benefits. Natural fisheries productivity can be increased, providing higher harvests at a lower cost (Lorenzen *et al.* 2001). Alternatively, fisheries enhancement could create new economic opportunities for fisheries-related industries (Lorenzen 2005), or help address the depletion of key fish stocks from overfishing (Brummett *et al.* 2013). Enhancement approaches are often socially preferably because they reduce the likelihood of applying additional unpopular restrictions under traditional management approaches (Grimes 1998, Borg 2004).

To be successful, techniques must contribute to a broad set of biological, economic, social and institutional management objectives, typically within complex fisheries systems (Lorenzen 2008). For fishery enhancement techniques to value-add, or outperform the alternative measures of traditional input-output controls, specific conditions may be necessary. Enhancement measures often encompass technical solutions that address natural or human-induced ecological limitations in natural systems to restore existing fisheries, increase productivity or develop new fisheries.

Manipulation of the environment or stocking of fish to provide or improve a fisheries resource is an ancient practice (Riggio *et al.* 2000, Seaman 2000). Since earliest times fishermen have sought ways to increase their catch or reduce the effort needed. Initial efforts focussed on building artificial reefs to attract fish and translocating fish to create new fisheries or to enhance existing populations. Since then, significant research and management effort has been invested trying to determine how this can be achieved most successfully and cost-effectively. More recently, greater emphasis has been placed upon the potential biological and non-target impacts of using fishery enhancements, including impacts to trophic food webs, genetics of wild stocks and large-scale stock depletion (Blankenship and Leber 1995, Lorenzen *et al.* 2010).

Fisheries enhancement can be classified into three broad categories.

1. Habitat enhancement (artificial reefs and fish attracting devices)
2. Fish stocking (stock augmentation, restocking, stock creation, ranching)
3. Habitat rehabilitation (rehabilitation of key habitats or ecosystem processes)

Each of these approaches have been employed by managers and fishers around the world in attempts to enhance fisheries outputs.

Fisheries enhancement has been utilised in all fishery sectors, from recreational, subsistence and artisanal fishers and commercial operations. The relative levels of use and the types of approach taken by different sectors varies between countries. For example, in Australia and the USA, fisheries enhancement has primarily focussed on improving recreational fishing. In contrast, Japan and South Korea have almost exclusively focussed their enhancement efforts on improving commercial fisheries. In Europe, the focus has been habitat protection of sensitive areas important to commercial fisheries, such as seagrass and other fish nurseries, whilst in many south-east Asian and Pacific Island countries, the focus of fisheries enhancement has been to improve subsistence and artisanal fisheries.

Within Australia, numerous enhancement approaches have been applied within a variety of fisheries, but broad and consistent uptake across different fishery sectors has been limited. Constraints to uptake include fishery manager knowledge levels and their ability to incorporate information on relative merits of different enhancement techniques into their fisheries management decision making processes. Quantitative comparison between their relative effectiveness of different approaches is often lacking. This has in part been due to the lack of quantitative socio-economic evaluations conducted on projects, but also because the decision to use fisheries enhancement techniques are often heavily influenced by politics. This is especially the case in the recreational fishing sector where enhancement projects are generally well received by the community and frequently derived from political promises.

1.1 Need

This project was identified by the SA Research Advisory Committee (SA RAC) as a FRDC priority in its January 2021 funding round, with funding commencing in October of 2021.

Despite a general trend for positive results from most fishery enhancement projects, not all approaches may deliver the best return on investment. Quantitative comparison of techniques is required to enable decisions to be made with greater certainty and to deliver the best value from an investment. A recent review into the value of man-made aquatic structures to fisheries by Harvey *et al.* (2021) concluded that understanding socio-economic values and benefits is a key component to guide any future decisions about fishery enhancement activities.

Broad uptake and application of some fisheries enhancement techniques by Australian fisheries managers has been limited. A major constraint has been the absence of clear comparative data on the relative costs and benefits for each approach and how they can be most effectively applied in different scenarios. Fisheries enhancement is widely practiced globally, and quantitative assessments exist for some techniques. Cost-benefit analyses have also been conducted for a few projects in Australia, but the results have yet to be consolidated and considered in the context of broader application by fisheries managers.

A consolidated fisheries enhancement knowledge base will enable robust comparisons of the relative return on investment for different approaches across various fisheries, and assessment of their long-term viability and impact on fishery productivity and sustainability. Such information will assist managers more clearly identify the most appropriate techniques to adopt and their potential benefits for their specific fishery, encouraging increased uptake and implementation. Clearer understanding of the relative merits and risks of different enhancement techniques will also enable appropriate techniques to be better incorporated into decision making processes, such as Harvest Strategies, and help identify critical knowledge gaps that need addressing.

“ When you can measure what you are speaking about, and express it in numbers, you know something about it; but when you cannot measure it, when you cannot express it in numbers, your knowledge is of a meagre and unsatisfactory kind; it may be the beginning of knowledge, but you have scarcely in your thoughts advanced to the state of science, whatever the matter may be.” Lord Kelvin (1889)

1.2 Objectives

A significant body of work has been undertaken on the impacts of fisheries enhancement techniques, but most evaluations have concentrated on the ecological outcomes, rather than the socio-economic value to fisheries. Ecological impacts can often be easier and cheaper to quantify, and promote our understanding of potential risks from applying enhancement techniques and how ecosystems adapt and evolve. However, ecological information is only one aspect in fisheries management decision making. Increasingly, it is recognized that fisheries should be managed for socio-economic objectives in addition to biological objectives (Radomski *et al.* 2001, Cowx *et al.* 2010). The focus of this review was therefore to synthesize information on the socio-economic costs and benefits of fisheries enhancement techniques to provide managers with a more comprehensive suite of data to inform their management decisions.

The objectives of this review were to:

1. Conduct a literature review of fisheries enhancement techniques, focussing on quantitative cost, benefits and socio-economic data
2. Conduct a cost benefit analysis to identify the efficiency of various enhancement techniques in different scenarios.

1.3 Framework for the review

This review consisted of multiple steps to ensure a comprehensive suite of appropriate data was collated and presented in a manner useable by fisheries managers. Initially, a systematic literature review was conducted to identify potential data sources. The review encompassed projects from both

Australia and worldwide, to ensure comprehensive coverage and to identify sufficient data for analysis. This information was combined into a cost-benefit analysis to compare the relative benefits and value of different enhancement techniques in different scenarios. Each of the three categories of fisheries enhancement were initially addressed separately, before this information was combined to provide a more holistic approach.

1.3.1 Literature review process

A systematic review of literature was undertaken to identify quantitative data on the costs, benefits and socio-economic evaluations of fisheries enhancement projects both within Australia and globally. As interest in using enhancement strategies in fisheries management has grown, several literature reviews and meta-analyses of the different fields of fisheries enhancement have been conducted, both globally and in Australia. A large component of published research discussed in these reviews focusses on the biological responses, such as changes in relative fish abundance and ecosystem evolution following fisheries enhancement activities. Although understanding of these processes is critical for artificial reef use in fisheries management and assessing the effectiveness of projects, the objective of the current review was to specifically focus on the cost, benefits and socio-economic impacts derived from such projects. Therefore, only a brief overview will be given on ecological outcomes, and readers are encouraged to refer to the reviews outlined in each Chapter for more detailed information on these topics.

This review covers three broad categories of fisheries enhancement techniques: habitat enhancement, fish stocking and habitat rehabilitation. Within each of these categories, project data was collated from both academic (peer-reviewed) and professional (technical) reports. All relevant articles were stored in a reference library in Endnote™. Literature was then screened based on three criteria: title, abstract and complete manuscript.

The literature search was initially conducted on articles from 1969 to 2022 that were indexed in English and examined any aspect related to the cost and benefits of fishery enhancement practices. Studies from all disciplines, ranging from economics, fisheries management, environmental management and rehabilitation, tourism and ecology were included. Only studies that provided data on the effectiveness, or costs and benefits of fishery enhancement techniques were included. Articles describing activities whose principal function was not to support fisheries development (*e.g.* breakwaters, shipwrecks, conservation stocking, offshore wind turbines) were not included. These activities may have some benefit to local fisheries, but would likely have been implemented in a different manner if fisheries enhancement was the objective. Thus it would have been inappropriate to include their values under this context. Journal articles and books published by reputable publishers were deemed as high-quality research, and therefore, included in the review. A significant number of technical reports also contained highly relevant data. These were included if they were published by a government agency, university or other reputable group, were of high-quality and well referenced.

Due to the broad scope of terms describing fishery enhancement projects, a substantial list of keywords was necessary. The databases were searched using synonyms for 'fisheries enhancement', 'habitat enhancement', 'fish stocking', 'habitat rehabilitation', cost-benefit' and 'socio-economic value'. As the various databases and search engines can yield different results (*e.g.* Calver *et al.* 2017), the literature search was conducted in the following leading databases:

- Scopus
- Web of Science

- CABI
- Wiley
- Science Direct
- Taylor and Francis
- Oxford
- Cambridge
- Springer
- CSIRO journals
- Canadian Science Publishing
- Google Scholar.

The citation of each publication from each search was downloaded and used to create a reference library in Endnote™. The search was conducted within the title, abstract and keywords using the terms related to the subject. Duplicate indexed articles were excluded and considered as only one document. Articles were also excluded if the complete document was not available online or accessible via inter-library loan. At first, the title of the literature was studied; if found relevant, then the abstract of the literature was carefully read. Finally, the full text of the literature was studied if the abstract was found relevant. All papers were given a relevance rating (1-5) in the Endnote database. Papers with a rating of two or less were not included in the final analyses.

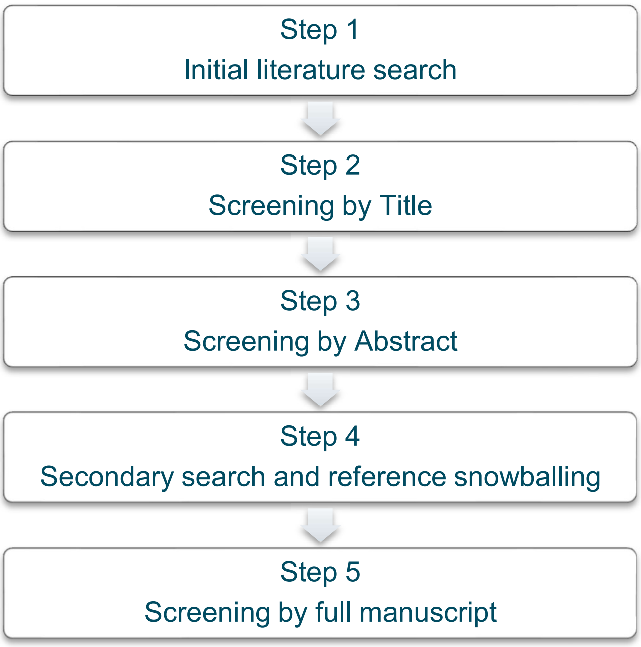

Figure 1 Literature review process

The initial search returned insufficient articles for some areas of interest. Therefore a secondary manual search was undertaken using a snowball technique analysing the reference lists of relevant studies and a broader internet search using the Google™ search engine. This approach identified a substantial number of additional relevant references, particularly in the professional literature (only published as technical reports).

From each article, information was extracted on the following four subtopics: (1) details of the fishery enhancement activities (*e.g.* location, scale, nature of activities undertaken, target species, fishery sector), (2) costs associated with implementation, (3) fishery benefits attributable to the enhancement activities, and (4) any socio-economic values or analyses.

A total of 2,303 citations were extracted from the initial and secondary searches. These were reduced to 1,869 when duplicate publications were removed, with 1,541 of these being accessible and had at least the abstract written in English. Screening by abstract further reduced this to 809 articles, whilst screening after reading the full manuscript left 224 articles. Of the three main categories of fisheries enhancement, habitat enhancement (83) contained the greatest number of relevant publications, followed by fish stocking (78) and habitat rehabilitation (63).

1.3.2 Socio-economic impact assessment and cost-benefit analysis

Most fishery management decisions involve quantitative choices: how many, what size, how large and area, how many fishers allowed, how much fishing effort, how much harvest etc. (Walters and Martell 2004). Decisions regarding fishery enhancement programs require information on the socio-economic as well as the ecological effects of enhancement activities. Fishery enhancements may often be quite expensive to undertake, and given they have the potential to cause over-exploitation in some scenarios, there is clearly no guarantee their economic effects will be positive. Since fishery enhancement is often undertaken on public interest grounds, decision makers need to understand whether such actions can be justified, and which approaches may most cost-effectively achieve the desired goals. Without a clear notion of what costs and benefits are likely to arise, there is a risk that fishery enhancement will be undertaken in circumstances which do not justify them. Therefore, economic appraisal of enhancement projects is an essential step in determining the role such strategies may play in fishery management and predict the economic worth of a project or management action.

Economic valuation and economic impact analysis are two widely used but distinctly different economic measures that can be used to evaluate and compare the economic outcomes from fisheries enhancement projects. The first provides information on the actual or potential impacts of a fishery enhancement project, whilst the second determines whether the project is an efficient investment. It is important to distinguish between these because they are commonly confused (Burgan and Mules 2001, Watson *et al.* 2007). More information was available on economic values; however the results from the literature review report both type of results where information is available.

The economic concept of value has been broadly defined as any net change in human well-being or welfare (Northern Economics Inc. 2009). Economic value measures the net economic welfare derived by society from policy or program changes (Wainger and Mazzotta 2011). The basic assumption of economic value is that the value of all goods and services can be expressed in an equivalent term of money and the value will be based on good utility contributions to humans (Eberle and Hayden 1991). Values can be differentiated into ‘producer’ and ‘consumer’ surplus. In terms of fisheries, producer surplus is the change in commercial fishermen’s net income or profits following a fishery enhancement project, and therefore equals total revenue (market price of total harvest) minus total costs (Milon 1991). Producer surplus can be improved by increasing the value of the harvest or decreasing harvest costs for any given amount of effort (Whitmarsh and Pickering 2000). By comparison, consumer surplus accrues to those who enjoy the outcomes of the economic activity and is a measure of the value of those goods or activities to the end user. It is the additional non-market willingness to pay (WTP) value over and above the expenditure on an experience (Whitmarsh and Pickering 2000). It a satisfaction or experiential value and thus not dependent upon the market value of the fish caught. In the context of this review, consumer surplus is measured by the recreational fisher's willingness to pay for fishery enhancement at a site. Economic values are also commonly presented in terms of various components which together make up the total economic value: direct use, indirect use and non-use values (see Table 1 for definitions).

Table 1 The different economic values associated with habitat enhancement using artificial reefs. Adapted from Whitmarsh *et al.* (2008).

Total economic value		
Direct use values	**Indirect use values**	**Non-use values**
Benefits arising from the immediate use of an artificial structure in the form of outputs that can be consumed or enjoyed directly	Benefits that an artificial structure provides to support other economic activities, or positive externalities that affect other users of the marine environment	Benefits from knowing that a marine asset has been conserved (existence and bequest motives) or may be available for use at a later date (option motive)
Examples: • Extractive uses (*e.g.* commercial and recreational fishing, off-shore aquaculture)	Examples: • Non-extractive uses (*e.g.* surfing and diving tourism) • Fish production via habitat protection (*e.g.* seagrass) • Effort diversion from overexploited fisheries • Coastal and shoreline protection • Water quality improvement via nutrient removal	Examples: • Knowledge that reef-based protection has increased marine biodiversity • Knowledge that a unique habitat is conserved intact for future generations

Satisfaction or enjoyment cannot be directly measured using market values. Instead they are typically calculated through revealed preferences (observations of actual behaviour) or stated preferences (inferred from questions). Two of the most commonly used approaches are the travel cost (revealed preference) and contingent valuation (stated preference) methods. The travel cost method assumes that travel and time costs incurred by recreational users reflects their willingness to pay for recreational enjoyment, and hence the value of that site/enhancement for that purpose (Whitmarsh and Pickering 2000). It should be noted that travel cost provides a lower bound estimate of economic value because it does not capture the additional value over and above what is paid *i.e.* consumer surplus (Whitmarsh and Pickering 2000). Conversely, contingent valuation is based on direct questioning to elicit the monetary value respondents place on an environmental amenity or resource (Parsons and Myers 2016). One advantage of contingent valuation is that resource users can be asked how much they would be willing to pay for implementation of new fishery enhancement projects before they are undertaken. Both of these techniques can produce similar estimates of the willingness to pay for different social groups, which can then be aggregated to calculate the economic value.

Economic impact on the other hand, measures the net change to the economy of a region that can be attributed to an activity, that would otherwise not have occurred had the activity not been undertaken (Watson *et al.* 2007). It provides decision makers with information on how policy changes affect economic activity, as measured in terms of output, jobs, income, or value added in communities, local government areas, or even at the state or national level (TCW Economics 2008). Economic impacts are generally measured using input-output (I/O) models, which describe and quantify the interdependencies between various sectors of producers and consumers that make up a regional economy (Miller and Blair 2009). Impacts can be direct (purchases of goods and services by consumers from producers), indirect (businesses buying and selling to each other) or induced (household spending based on the income earned from the direct and indirect effects) (Northern Economics Inc 2009, Cook and Becker 2016). Induced impacts are calculated through regional economic multipliers. The total economic impact of consumer spending is equal to the sum of all of these impacts.

Two of the main economic assessment approaches are economic impact analysis and cost benefit analysis. The first provides information on the actual or potential impacts of a habitat enhancement project, whilst the second determines whether the project is an efficient investment. The primary advantage of cost benefit analysis is that it can be used to express a broad array of benefits received and costs incurred by different social groups in a single monetary measure.

For public sector projects (such as fishery enhancement), an economic appraisal using cost benefit analysis is typically more appropriate than a financial appraisal using impact economic analysis because both externalities (costs and benefits) and intangible impacts need to be considered (Whitmarsh and Pickering 2000). Cost benefit analysis compares the impact of an activity to that of the base case (*i.e.* continue doing nothing approach) which represents the minimum cost of using the existing arrangements to deliver services at current levels and standards. When comparing the cost of alternative options, it is important to consider 'whole-life' costs. These are costs incurred from the outset of the initiative throughout the expected life of the project. Given that fishery enhancement can be costly to undertake, and significant public investment is often associated with projects, the question becomes whether society as a whole is expected to be better or worse off if the project is undertaken. The values based approach of cost benefit analysis is therefore more appropriate to use because it considers both the costs and benefits to determine whether the monetary value of the outcome justifies the cost (Milon 1991).

Cost benefit analysis focusses only on changes to costs and benefits resulting from a particular action (*i.e.* marginal values) to determine if the result has added value and counts only the direct impacts on income that result from a project. This avoids the problem of double-counting benefits and provides a consistent basis to compare benefits and costs (Milon 1991). The greatest value provided to fisheries from enhancement projects generally accrue to one or more user groups that can be described collectively as commercial fishermen, recreational anglers and non-extractive users (divers, snorkellers, kayakers etc). Values for non-users are also considered, but are much harder to estimate. Cost benefit analysis enables the values for these groups to be estimated either individually or collectively and is especially useful for comparing between different types of fishery enhancement projects (Bishop *et al.* 1990). Cost benefit analysis therefore makes it possible to determine whether a proposal has a net benefit and which of the alternative proposals (including no-change) has the greatest net benefit. It provides a clear framework for weighing up different options and determining investment priorities across diverse sectors.

Cost benefit analyses commonly use three simple socio-economic indicators to help compare and rank different options (Whitmarsh and Pickering 2000). The most basic is the project's net present value (NPV), which equals the present value of all benefits minus the present value of all costs over the life of the project. If the NPV is greater than zero, then the project is deemed to have a net benefit and would have economic justification to proceed. When comparing projects, the larger the NPV, the greater the benefits. This measure is useful to compare projects when there are unlimited budgets to invest with as it shows which project will deliver the greatest absolute benefit (Rogers *et al.* 2018). The second metric is the benefit cost ratio (BCR), which is the ratio of the present value of benefits to the present value of costs. The ratio must exceed one for the proposal to generate a net benefit. The larger the ratio, the greater the benefit generated per dollar invested (Rogers *et al.* 2018). This measure is more useful for prioritising projects when budgets are limited. The third metric is the internal rate of return (IRR), which is the discount rate at which the net present value of a new investment's expected costs and benefits equals zero. Use of IRR was rare in the reviewed literature and as such is not included in our analysis. It should be noted that although both NPV and BCR will provide the same positive or negative outcome for an alternative, where various options are considered, the two methods will not always give the same preferred outcome (Whitmarsh and Pickering 2000). Presenting both sets of results may therefore be most appropriate to provide decision makers with the most information with which to make their final decision.

The costs incurred undertaking fishery enhancement and the benefits generated, arise over a period of time. This characteristic is of central importance, given that the biological processes underlying fisheries productivity do not occur instantaneously and that the physical life of the enhancement activities may extend over several decades (Whitmarsh and Pickering 2000). These costs and benefits need to be discounted to reduce future values to bring them into line with today's values. The discount rate selected can play a large role in the determining outcomes of cost benefit analyses (Whitmarsh and Pickering 2000). Where the there is uncertainty of the discount rate’s value or to evaluate the sensitivity it has on the outcomes, results from a range of discount rates can be compared. In the studies reviewed, social discount rates in the economic analyses ranged between 0% and 10%, with most between 3-7%. In Australia, discount rates between 4-7% are commonly used for environmental resource projects, but there is no single definitive value (Walker *et al.* 2008, Hone *et al.* 2022). Therefore we selected a rate of 5% for our cost benefit analyses, which was the median value from those used in the reviewed studies and matches those previously used in Australia.

The aim of this review was not to evaluate the total economic value for enhancement activities, but instead to investigate how the costs of different approaches compare with the resultant benefits to fisheries. Investment in enhancement projects for fisheries development is undertaken to improve fisheries value and sustainability. Additional socio-economic benefits will value-add to an enhancement approach, but are likely to be secondary considerations for fisheries managers when determining activities that will be undertaken. Therefore, this review focussed only on benefits accruing for fisheries from enhancement (*e.g.* van Vuuren and Roy 1993). As long as the net benefits to fisheries included in the analysis exceeded the net costs of implementation, the enhancement activity will provide net benefits from a fisheries management perspective (Krutilla and Fisher 1985).

In total, 224 articles examining socio-economic benefits of fisheries enhancement approaches were analysed to identify the country and year of study, enhancement type, the measured value types, valuation method and the valuation context or question. To enable accurate economic comparison between studies, all economic value estimates were converted to 2021 AUD values using the Reserve Bank of Australia’s historical currency conversion data for the relevant countries (available at https://www.rba.gov.au/statistics/historical-data.html#exchange-rates) and the Consumer Price Index published by the Australian Tax Office (https://www.ato.gov.au/rates/consumer-price-index/).

In this review, economic impacts and economic valuations resulting from fishery enhancement studies were reported separately for each of the three main enhancement categories. Where sufficient information was available, data within each category was further grouped to provide an overview of socio-economic benefits for various actions or scenarios (*e.g.* estuarine artificial reefs, offshore artificial reefs, freshwater artificial reefs). The data grouping facilitated comparison between activity types and the environments where they were undertaken. For each group where sufficient cost and benefit information was available, a cost benefit analysis was conducted using mean or median values, based on a 30 year time horizon and 5% discount rate. Where necessary, delays in the full realisation of benefits was incorporated to account for cumulative biologically driven increases in productivity. The results from the cost benefit analyses were used to compare the indicative NPV and BCR between groups and techniques, to help inform managers of the relative potential for each approach.

Limited primary socio-economic data has been collected on fishery enhancement activities in Australia. Therefore, global data was included in the analyses and comparison. While some of the costs and benefits values of fishery enhancement activities from other countries can be generalised and transferred to Australia, the usefulness of the information depends on the location specificity (*i.e.* local fine scale location specific areas of interest) required by proponents or decision makers (Harvey *et al.* 2021).

Chapter 2. Habitat enhancement

2.1 General introduction

The availability of habitat is an essential requirement for fish to accomplish daily and seasonal survival tasks such as foraging, sheltering and reproducing (Robertson and Duke 1987, Watson *et al.* 1993, Bloomfield and Gillanders 2005, Sheaves *et al.* 2007). The critical nature of habitat to fish and the often naturally limited availability of some key habitat types, means that augmenting existing natural habitats with artificial habitats has potential as a management tool to enhance fish abundance and productivity in habitat-limited areas.

Habitat enhancement has been widely used to manipulate fisheries resources for thousands of years (Sato 1985, Riggio *et al.* 2000, Seaman 2000, Ito 2011). The practice of installing structures to create artificial environments has probably been used since the Neolithic period by African peoples that noticed a greater abundance of fishes near floating and submerged objects (Ito 2011). There is growing attention on its how its use can be conducted more safely and effectively to increase the value generated from fisheries resources. This Chapter will provide an overview of habitat enhancement practices, discuss some of the challenges and opportunities involved with its application for fisheries management, and summarize the socio-economic costs, benefits and outcomes from past projects.

The term “habitat enhancement” has been used as a broad phrase encompassing various levels of intervention in relation to habitats. In general, the term is applied to actions that aim to improve the nature, quality, size or geographic distribution of a habitat type. Gwin *et al.* (1999) defined habitat enhancement as "the modification of specific structural features of an existing habitat to increase one or more functions based on management objectives". Following this train of reasoning and with specific regard to fisheries, in this review habitat enhancement is defined as “the deliberate and strategic placement of materials or structures into aquatic environments to create or enhance habitat for fish or fisheries activities”. This term excludes artificial islands, cables, pipelines, platforms, mooring and structures for coastal defence (*e.g.* breakwaters and dykes) which were primarily constructed for other purposes and wrecks that are accidentally present on the sea bed. Rehabilitation of aquatic habitats through the installation or recovery of biogenic habitats (*e.g.* oyster reefs, mangroves, seagrass etc.) will be covered in Chapter 4 on Habitat Rehabilitation.

Habitat enhancement in aquatic environments can typically be separated into activities undertaken in two distinct zones: on the benthos with artificial reefs, or within the mid to upper water column using suspended fish attracting devices (FADs). Artificial reefs are primarily aimed at demersal species whilst FADs predominantly target pelagic species.

Artificial reefs can be defined as consisting of one or more objects of natural or human origin purposely deployed on the seafloor to influence physical, biological or socio-economic processes related to living marine resources (Seaman 2000). They can vary greatly in their shape, structure, purpose and functionality and have been popular fisheries enhancement tools because fish biomass often increases in surrounding waters after their deployment (Bombace *et al.* 1994, Bortone *et al.* 2011, Leitão 2013, Champion *et al.* 2015, Miranda 2016).

Structurally complex shelter is scarce in most flat sedimentary environments, and often rendered more so by anthropogenic activities, such as the impact of swept bottom fishing gear which can destroy the cover of epifauna or epiflora (Caddy 2007). An ecological justification for artificial reefs is that the area and condition of reef structures may limit the abundance of fish, so increasing reef habitat using artificial reefs may increase fish production (Bohnsack 1989, Boretone *et al.* 2000, Champion *et al.* 2015). The installation of materials to form artificial reefs has been used to address this issue by increasing structural benthic habitat complexity. Artificial reefs can be attractive to demersal species because they provide suitable shelter from predators in the areas where habitat complexity is relative patchy and scarce (Schmid 2000).

Habitat enhancement can also be utilised in open water environments through the use of FADs. FADs can be defined as floating or suspended structures purposefully installed to attract fish. Structural complexity is very limited in open water environments and many fish species naturally congregate near objects floating in open waters (Kingsford 1993, Castro *et al.* 2002). Floating or suspended objects can provide shelter, feeding and reproductive opportunities for fish and are utilised by different life stages of many pelagic species (Parin and Fedoryako 1999, Castro *et al.* 2002). The propensity for fish to aggregate around floating structures has been well exploited through the use of FADs within recreational, artisanal and commercial fisheries. Globally, the justification for the development of many FAD programs has been to shift fishing pressure towards pelagic fish in areas where bottom fish were over-exploited or to diversify regional or national fishing activity (Taquet 1998, Kakuma 2000). FADs can be anchored or drifting. However drifting FADs can have significant environmental consequences (Marsac *et al.* 2000, Essington *et al.* 2002, Dempster and Taquet 2004) and their use in purse seine fisheries is not permitted in Australia (AFMA 2022). Domestic FADs are anchored, monitored and maintained by state authorities, whilst FADs aren’t currently used in Commonwealth managed fisheries.

As interest in using habitat enhancement in fisheries management has grown, a number of literature reviews and meta-analyses of the field have been conducted, both globally and in Australia. Matthews (1985), Kerr (1992), Branden *et al.* (1994), Jensen *et al.* (2000), Coutin (2001), Bortone *et al.* (2011), Paxton *et al.* (2020), Ramm *et al.* (2021) and Vivier *et al. (*2021) all reviewed global trends in artificial reefs and their potential role in various aspects of fisheries management. More recently, Diplock (2011), Becker *et al.* (2018), Florisson *et al.* (2018) and Harvey *et al.* (2021) have reviewed the use of artificial reefs and other man-made structures, with particular reference to their application in Australia. A large component of the published research discussed in these reviews focusses on the biological responses, changes in relative fish abundance and ecosystem evolution following reef installation. Although thorough understanding of these processes is critical in assessing the effectiveness of habitat enhancement projects, the objective of the current review was to specifically focus on the costs, benefits and socio-economic impacts that are generated. Therefore only a brief overview will be given of ecological impacts/outcomes, and readers should refer to the above reviews for more detailed information on these topics.

The costs, benefits and socio-economic outcomes of habitat enhancement projects are influenced by a wide range of variables, but key factors include the project objectives, structure design, scale of the activity and location. The role of these factors will be discussed in the following sections and socio-economic data from past projects synthesized to enable comparison between the effectiveness of different approaches.

2.1.1 Objectives of habitat enhancement

Habitat enhancement in aquatic ecosystems is undertaken for a variety of environmental, commercial, recreational and scientific purposes. Projects often provide benefits to multiple areas, but the primary objectives determine the specific nature of the activities undertaken and strongly influence the ultimate outcomes of a project. Clearly outlining project aims enables activities to be better defined and their effectiveness and impact to be better evaluated. In fisheries management, common goals for habitat enhancement projects include increasing fisheries production, increasing fisheries efficiency, reducing conflict between different resource users, providing new or improved recreational opportunities, mitigating habitat loss or the restoration of habitats (Becker *et al.* 2018). However, the most prominent uses for habitat enhancement in fisheries management have been to attract mobile species to structures to increase harvest efficiency or to increase the long-term local fish biomass (Paxton *et al.* 2020).

Habitat enhancement aims to increase fishery value and sustainability through management actions targeted at various key biological processes which govern fishery stocks. At the biological level, the objectives typically include increasing survival, growth, shelter or food resource availability (Bortone *et al.* 2011). This can be achieved in a number of ways, including through provision of protection from predators (Bohnsack and Sutherland 1985, Santos and Monteiro 2007, Becker *et al.* 2018), increasing access to breeding sites and mates (Hunt *et al.* 2002), provision of nursery habitat (Fabi *et al.* 2015, Mercader *et al.* 2017), and increasing food production, diversity and access for target species (Cresson *et al.* 2014, Granneman and Steel 2014, Champion *et al.* 2015).

The socio-economic objectives of projects are often less clearly outlined, although there is a growing trend for their specification and consideration in modern fishery enhancement projects. Habitat enhancement can directly or indirectly influence harvest and access costs, fishery accessibility, net income, fisher success, experience and perceptions, value per unit effort and the flow-on economic values for communities associated with these parameters (Ramos *et al.* 2008, Whitmarsh *et al.* 2008, Seaman *et al.* 2011). Significant differences in the socioeconomic objectives are often observed between projects targeting commercial and recreational fisheries.

Habitat enhancement can aim to attract fish, increase natural production or a combination of both. There is a sliding scale between the relevant importance of the two objectives and they are rarely mutually exclusive. For many habitat enhancement projects, this is seldom explicitly outlined at the start, as objectives are often listed as harvest improvement, with limited details given on how this is to be achieved. The type of fishery has a strong influence on project objectives due to the relative differences in their scale of harvest and how this affects the fishery stocks. Recreational fisheries are often regarded as having less intense localised harvest pressure and attraction of fish to an artificial structure can often be managed acceptably within existing management frameworks (Diplock 2011). Conversely, the higher harvest rates in semi-intensive and intensive commercial fisheries can have a greater impact on localised fish stocks (Grossman *et al.* 1997, Lima *et al.* 2018). This requires a greater emphasis to be placed on productivity improvements to ensure long-term sustainability. The impacts of small-scale and artisanal fisheries typically lay somewhere between these, and the objectives can vary with socio-economic factors (Jensen *et al.* 2000). In developing countries attraction of fish to traditional grounds has been the primary focus of habitat enhancement projects (*e.g.* Garcia 1991, Islam *et al.* 2014), whilst in more socio-economically developed regions consideration of productivity improvements takes on a greater importance (Polovina and Sakai 1989, Kim *et al.* 1994, Ito 2011).

Projects which focus specifically on fisheries harvest aim to increase the exploited fish biomass or harvest efficiency by increasing the desirable habitat of target species. The goals are typically to achieve a relatively quick ecological response and benefits for the fishery. This may involve the creation of habitat in regions where there previously was none, through the creation of refugia, spawning areas, and food resource availability (Spieler *et al.* 2001, Jaxion-Harm and Szedlmayer 2015). The scale of such projects can be highly variable because increased production is not the primary objective, and project size often reflects the scale of harvest pressure in the target fishery. Recreational and artisanal fisheries may be able to utilise cheaper and smaller enhancement activities because they often have a smaller impact on fisheries resources, whilst more sizable extractive commercial fisheries may require larger and more costly habitat enhancement.

Employing habitat structures to restore or improve natural production occurs by creating additional habitat and food resources that address identified bottlenecks for target species (Leitao *et al.* 2013, Champion *et al.* 2015, Mercader *et al.* 2017). This can occur in areas where habitat has been degraded, is limited or where habitat modification could lead to better fisheries production. The scope of such projects is dependent upon the scale at which degradation has occurred or the level of improvements desired. Typically the goal involves long-term improvement of regional fisheries resource production. If significant increases of productivity are desired, large areas may require enhancement and the ecological response time may be longer (Santos *et al.* 2007, Bortone *et al.* 2011, Ito 2011, Becker *et al.* 2018, Lima *et al.* 2020).

2.2 Trends in habitat enhancement

2.2.1 Global trends

The use of habitat enhancement as a fisheries management tool is an ancient practice (Riggio *et al.* 2000, Seaman 2000, Ito 2011). Throughout history, artificial reefs and fish attracting devices made of different materials have been used in more than 50 countries with the aim of attracting fishes (Polovina 1991, Diplock 2011, Langhamer 2012). Their use is now globally widespread, although they exhibit marked regional and national differences, reflecting to a large extent the wider policy role that these structures are expected to fulfil (Becker *et al.* 2018).

Japan, China, Taiwan and South Korea have almost exclusively focussed their artificial reef efforts on improving commercial finfish, gastropod, marine algae and shellfish fisheries. These countries use large-scale habitat enhancement to restore, enhance or create new commercial fishing opportunities. As recreational fishing has gained popularity, greater use of some of the installed structures is starting to occur and reefs specifically targeting the recreational sector are also now being installed (Chen *et al.* 2013, Seung and Kim 2018).

Japan is one of the most prolific and successful exponents of habitat enhancement and has installed more than 20 million m^3 of artificial reef at 6400 sites (Barnabe and Barnabe-Quet 2000). The government has invested billions of dollars and is at the global forefront of using habitat enhancement in fisheries (Polovina and Sakai 1989). The Japanese program uses specially designed modules in both shallow and deep waters to increase commercial production of a wide range of species, including pelagic and demersal fish, squid, abalone, marine algae, oysters, lobsters, sea cucumbers, octopus and sea urchins.

South Korea also invests heavily in habitat enhancement. In 2011 more than $55 million p.a. was being invested into their artificial reef program, with over 270,000 ha of reef installed (Department of Fisheries 2010). Commercial shellfish, crustacean and seaweed cultivation occurs using concrete structures in shallow waters and finfish production occurs on larger steel and concrete structures in deeper waters (Kim *et al.* 1994).

In Europe, the primary focus of artificial reefs has been to prevent illegal trawling in sensitive areas, such as seagrass and other fish nurseries. Here, artificial reefs are seen as a management tool for sustaining coastal fisheries and compensating for the effects of stock depletion, a role made all the more important where artisanal fishing supports livelihoods and helps maintain the integrity of local communities. For example, in France more than 80% of reefs have been installed with the primary objective of protecting artisanal fisheries (Jensen 2002, Fabi *et al.* 2011) and approximately half are located in protected areas (no-fishing) to act as sources of increased productivity for the region (Fabi *et al.* 2011, Tessier *et al.* 2014).

Similarly, in African (Lechanteur and Griffiths 2001, Seaman, 2002, Omofunmi *et al.* 2018), Middle Eastern (Feary *et al.* 2011, Eighani *et al.* 2019), south-east Asian (Garcia 1991, Munro and Balgo 1995, Waltemath and Schirm 1995, Yusfiandayani *et al.* 2013) and Pacific Island (Albert *et al.* 2014, Bell *et al.* 2015, Tilley *et al.* 2019) countries, artificial reefs and fish attractors form an important component of artisanal and small-scale fisheries in both marine and freshwater fisheries. The major commercial application has been the use of FADs for the tuna purse seine fishery (Sharp 2011).

In contrast, the focus of habitat enhancement in Australia and the USA has primarily been on improving recreational fishing (Kerr 1992, Diplock 2011, Florisson *et al.* 2018). The USA has installed more than 1 million m^3 of artificial reefs at over 1500 marine sites (Barnabe and Barnabe-Quet 2000, Sutton and Bushnell 2007, Ropicki *et al.* 2021) and habitat enhancement is utilised in lakes and reservoirs by more than 80% of state fisheries agencies (Tugend *et al.* 2002, Miranda 2016). Few reefs have been installed specifically for commercial fisheries, although the red snapper fishery in the Gulf of Mexico is indirectly supported by artificial reefs and the habitat created by the abundance of oil and gas infrastructure (Gallaway *et al.* 2009, Shipp and Bortone 2009).

2.2.2 Habitat enhancement in Australia

Development of habitat enhancement for fisheries management in Australia has followed a similar trajectory to global trends, but historically lagged behind. The focus in Australia has primarily been for improving recreational fishing, although several trials have been conducted for commercial marine invertebrate fisheries and offshore pelagic fisheries.

Like elsewhere, early habitat enhancement occurred using materials of opportunity, generally in low energy environments with limited natural hard structure (Pollard 1989). For example, by 2001, there were at least 106 artificial reefs reported in Australia, of which 37% were made from tyres, 22% from scuttled vessels, and only 6% from concrete (Coutin 2001). The focus was on providing structures to aggregate fish for recreational anglers (Kerr 1992, Diplock 2011). Projects were typically conducted in an ad hoc rather than strategic manner and monitoring and evaluation was limited (Diplock 2011, Florisson *et al.* 2018). The materials of opportunity used in some projects have since broken up, buried or had to be removed due to leaching contaminants into the water leading many governments to ban the use of materials of opportunity for artificial reef construction (Diplock 2011, Bowen *et al.* 2020). More detailed descriptions of historic habitat enhancement projects in Australia can be found in the comprehensive reviews conducted by Kerr (1992), Diplock (2011) and Florisson *et al.* (2018).

Since 2001, there has been a surge in interest and application of habitat enhancement projects. This may be partly due to the increased availability of funding for such projects, due to recreational license fees and political promises, greater recognition of the economic value of recreational fishing (*e.g.* Henry and Lyle 2003, Ryan *et al.* 2013, West *et al.* 2013, West *et al.* 2015, BDO 2021), and also the positive results reported from the more rigorous evaluation processes conducted as part of modern projects installing purpose-built reefs. Artificial reefs and FADs have also shown great popularity with recreational fishers and anglers, who actively advocate for further expansion.

The focus for habitat enhancement has now strongly shifted towards purpose-built structures, with installations now used across Australia in coastal waters, estuaries and bays. Major programs installing purpose-built artificial reefs and FADs for recreational fisheries have been established in New South Wales, Western Australia, Victoria, Queensland and the Northern Territory. Limited commercial application currently occurs, although research is underway on how habitat enhancement can benefit the commercial sector.

Pilot studies in New South Wales and Victoria evaluated the effectiveness of small concrete habitat modules (Reef Balls™) in estuaries and bays . In 2004, NSW secured funding through recreational fishing licence fees for the deployment and monitoring of artificial reefs in several estuaries which had been declared recreational fishing havens. Monitoring of the pilot program demonstrated the benefit of these reefs through sustained recruitment of species which were highly regarded among recreational anglers (Lowry *et al.* 2010, 2014, Becker *et al.* 2017) and the program expanded to include a total of nine artificial reefs located within six estuaries. Following the success of the estuarine artificial reefs, the program was expanded to examine three large offshore artificial reefs located in shallow waters accessible by trailer-boat recreational anglers. The positive results from these concrete and steel pilot reefs (Lowry *et al.* 2015, Becker *et al.* 2017), has seen the program expand to a total of nine sites spread along the entire NSW coast, and more are planned. FADs were first trialled in NSW in the 1980's but the expense was initially too high to justify their continued use. With the advent of more economical designs, NSW established a dedicated recreational fishing FADs program in 2002. This program now has a total of 32 FADs currently deployed and maintained from Tweed Heads in the state's north to Eden in the south.

Victoria was one of the first states to trial artificial reefs for commercial fisheries. Between 2000 and 2003, stocking abalone seed onto artificial reefs in Port Phillip Bay for commercial abalone ranching was trialled. The artificial reef design was found to be of sub-optimal design and the location was deemed only marginally viable for ranching activities (James *et al.* 2007). The trial of artificial reefs for recreational fishing enhancement in Victoria commenced in 2008 in Port Phillip Bay with reefs constructed from concrete habitat modules (Reef balls™) deployed at three sites for boat-based fishing. Based on the outcomes of the trials (Hamer 2006), eleven additional reefs have been added around the state, mostly using concrete modules. Three of the reefs consisted of rock and limestone rubble seeded with oysters and mussels. The reefs have been located to provide access to a mix of shore and boat-based anglers. A purpose-built offshore artificial reef, similar to those used in NSW, was installed near Torquay in 2015 and planning for an additional reef in Port Phillip Bay is underway. Five FADs have also been deployed along the Torquay artificial reef to attract pelagic game fishing species for anglers. The deployments of these FADs follows on from a FAD trial that was undertaken at two sites in 2007-2008.

Artificial reefs for fisheries enhancement in Western Australia were initially limited to a commercial ranching. Research in the 1970's examined artificial reef ranching as a possibility for the western rock lobster fishery, but the trials were unsuccessful, and the practice not adopted (Pollard 1989). More contemporary research has continued to investigate the viability of this approach (Phillips *et al.* 2007). Currently abalone ranching occurs near Augusta and has demonstrated benefits for the abalone industry (Melville-Smith *et al.* 2013, Melville-Smith *et al.* 2017). More recently Western Australia has invested considerable research into the use of artificial reefs and FADs for recreational fisheries. Six artificial reefs have been deployed along the Western Australian coast, from Esperance to Exmouth, using purpose-built concrete modules and steel structures. In 2018, the Exmouth Integrated Artificial Reef (King Reef) was deployed and was the first installation in Australia to combine repurposed oil and gas infrastructure with purpose-built concrete reef modules (Florisson *et al.* 2020). Significant research has been undertaken into how oil and gas infrastructure from the North-West Shelf could be utilised to enhance fisheries. Research has also been conducted into the use of artificial reefs in impoundments to boost marron survival and fishery productivity (Molony and Bird 2005, Beatty *et al.* 2019). Besides artificial reefs, more than 32 FADs are in use across the state to aggregate pelagic species for recreational anglers. Historically most of the FADs were installed and maintained by game fishing clubs. However, the majority are now operated by the state government. A pilot program was undertaken in the early 1980's using FADs for the commercial tuna and shark fisheries in southern waters. In the initial trials, 34% of commercial tuna harvest in the region came from the FADs, but varying levels of success were observed as the program expanded (Pollard 1989).

In the Northern Territory artificial reefs have long been established in Darwin Harbour using scuttled vessels, mining trucks, mixed materials of opportunity and concrete pipes and culverts (Diplock 2011). In 2019, four specifically engineered concrete reef complexes and seven FADs were installed to improve recreational angling. No habitat enhancement has been undertaken for commercial fisheries.

Artificial reefs for fishing based on materials of opportunity (particularly tyres) and scuttled vessels have a long history in South Australia (Diplock 2011). Approximately 19 large reefs have been installed in the state, however, numerous smaller private reefs were also illegally created by anglers hoping to create private fishing hotspots in the State's gulfs. Modern, purpose-built artificial reefs have yet to be utilised. A large wave energy generator that sunk during transport is proposed be cleaned and repurposed into a reef for fishing and diving.

Tasmania also has no large purpose built artificial reefs for recreational fishing, but two artificial reefs, consisting of pre-cast concrete modules, are planned for deployment on the east coast (Fisheries Tasmania 2022). Two trial artificial reefs had been installed in an estuarine marine reserve to examine the viability of rock lobster ranching on artificial reefs, but no additional reefs have been installed for this purpose. In 2021 three FADs were deployed off Tasmania's east coast as part of a trial program. This has now expanded to five FADs (Fisheries Tasmania 2022).

Habitat enhancement for recreational fisheries in Queensland waters using materials of opportunity and scuttled vessels also have been used for a long time (Diplock 2011). Several extensive reef complexes were created in the partially sheltered waters of Moreton Bay and Hervey Bay from the 1960's onwards. More recently the Queensland Government established seven purpose-built artificial reefs in Moreton Bay Marine Park, and enhanced an existing artificial reef by scuttling a large vessel to provide recreational anglers with a range of fishing opportunities (DES 2022a). A further two concrete module reefs were also established in Hervey Bay in the Great Sandy Marine Park in 2015 (DES 2022b). These reefs were created in order to partially offset recreational fishing opportunities lost through the rezoning of the Marine Parks. A pilot program has also been implemented to evaluate the use of fish attracting structures in stocked freshwater impoundments. Initial results indicate that the local abundance of several stocked fish species increased following the addition of habitat structures consisting of organic materials, purpose-built and suspended designs (Norris *et al.* 2021). The deployment of FADs in Queensland for recreational fishing has increased significantly over the last decade. Currently 32 surface FADs, 4 all-water FADs and 12 subsurface FADs have been installed along the Queensland coast, with additional structures being considered. Investment in FADs increased to encourage anglers to target primarily fast growing, short lived pelagic species to increase recreational fishery diversity and relieve pressure on demersal fish stocks.

2.3 Impacts of habitat enhancement

2.3.1 Ecological impacts

Habitat enhancement can deliver a variety of ecological benefits for fisheries. Many fish populations are limited to available complex habitat (*i.e.* hard substrata) and therefore an increase in habitat (*e.g.* via the installation of an artificial reef or FAD) is likely to increase the carrying capacity. This can lead to increases in the ecological productivity and biomass of the area. Habitat enhancement can help address ecological bottlenecks based on habitat limitation, potentially increasing survival, spawning, recruitment and growth (Bohnsack and Sutherland 1985, Santos and Monteiro 2007, Bortone *et al.* 2011, Fabi *et al.* 2015, Mercader *et al.* 2017, Becker *et al.* 2018). This can result in significant increases in fish assemblages at a site, but may also change the community composition (Cresson *et al.* 2014, Granneman and Steel 2014, Champion *et al.* 2015). At sufficient scale, habitat enhancement also has the potential for regional benefits, particularly if the fish populations at the new habitat act as a source and result in a net emigration into other nearby areas (Relini *et al.* 2002, Roa-Ureta *et al.* 2019, Becker *et al.* 2022).

2.3.1.1 Diversity, biomass and abundance

A range of descriptors for the aquatic community, such as fish abundance, species richness, biomass and diversity, are often used to describe changes in fisheries productivity related to habitat enhancement (*e.g.* Scott *et al.* 2015, Stevens *et al.* 2018, Streich *et al.* 2018, Lima *et al.* 2020). The assessment of these ecological descriptors has helped evaluate the potential use of habitat enhancement structures in attracting and potentially contributing to production of fish species of exploitable interest (Macusi *et al.* 2017, Lima *et al.* 2020).

The additional habitat provided by artificial reefs has the potential to increase the environmental carrying capacity across a range of trophic levels (Gratwicke and Speight 2005). The biomass of benthic invertebrates has been found to significantly increase around habitat enhancement structures. (*e.g.* Sampaolo and Relini 1994, Dance *et al.* 2011). The sessile fauna and algae that develop, serve to attract fish by increasing heterogeneity in the structure's topology (Brotto and Zalmon 2007), and by providing essential food sources and shelter (Wallace and Benke 1984, Leitao *et al.* 2007). These fish in turn attract larger higher trophic level species (Champion *et al.* 2015, Becker *et al.* 2017, Stevens *et al.* 2018).

Large shifts in the structure of fish assemblages are commonly observed following the deployment of artificial reefs (Bohnsack *et al.* 1994, Leitão *et al.* 2008, Thanner *et al.* 2006, Lima *et al.* 2020). When deployed in areas with bare and homogeneous substrates, species richness typically increases substantially (Santos and Monteiro 1997, 1998, Santos *et al.* 2007, Lowry *et al.* 2010, Folpp *et al.* 2013). Lima *et al.* (2020) observed species richness and diversity to almost double at nearshore sites with large artificial reefs, compared to nearby bare control areas. However, more inconsistent results have been observed when these values are compared between natural reefs and artificial reefs. Even where species richness and diversity values are similar, the structure of community assemblages often differs between bare reference sites, natural reefs and artificial reefs (Cresson *et al.* 2019). Identifying the potential differences and ascertaining the reasons why, helps understand the drivers behind the fish community structure and can be used to evaluate and enhance the design of artificial reefs to improve the benefits to fisheries (Carr and Hixon 1997).

The structural complexity of artificial reefs rarely mirrors that present at nearby natural reefs, resulting in differences in the associated fish communities (Carr and Hixon 1997, Folpp *et al.* 2013, Zalmon *et al.* 2014). The designs of modern purpose-built reefs structurally differ considerably from natural reefs, whilst still incorporating features (*e.g.* void spaces and towers) aimed at providing habitat for a range of species. Consequently, fish assemblages associated with artificial reefs may not directly mimic the assemblages found at natural reef sites (Folpp *et al.* 2013, Becker *et al.* 2017). This is especially the case for high profile marine structures which attract both benthic and pelagic species (Champion *et al.* 2015, Becker *et al.* 2017). Such shapes rarely occur in natural reefs. Folpp *et al.* (2013) found that concrete reef modules deployed in three New South Wales estuaries all harboured a different fish assemblage to nearby natural rocky reef and bare sand sites. Species richness and abundances were greater on the artificial reefs, which were attributed to the reef harbouring its own endemic fauna, as well as supporting populations species found on the adjacent natural habitats.

The abundance of exploitable fish species can be higher at the artificial reefs, whilst lower trophic order grazers are more dominant at natural reef assemblages (Folpp *et al.* 2013, Lima *et al.* 2020). This particular trait creates potential for fisheries management to drive changes in community assemblage that improve the yield and catch value of specific desirable species through the introduction of well-designed artificial reefs.

The invertebrate and fish community assemblage at artificial reefs is not static and undergoes a long period of evolution prior to forming stable communities (Coll *et al.* 1998, Relini *et al.* 2002, Zalmon *et al.* 2014, Cresson *et al.* 2019). Successional processes on and around the structure drive this change. Early colonization by benthic-feeding, reef associated species is likely a function of the shelter the artificial reef provides, rather than other ecological functions such as a food source. This is because the development of sessile communities takes time whilst structural habitat availability can be immediate (Relini *et al.* 1994). This means that there may be a lag period before the benefits from artificial reef installation are fully realized. Successional change also means the impacts will potentially improve over time.

Many studies monitoring fishes on artificial reefs have relatively short duration (1-3 years), and the need for longer-term studies has been identified in a number of global reviews (*e.g.* Baine 2001). Short-term studies may be ineffective in detecting prolonged fluctuations or disturbances in ecological systems, leading to bias in conclusions about the performance of habitat enhancement. In contrast, long-term studies can provide the necessary understanding of fish assemblage variation, revealing subtle but consistent biological trends which could be beneficial in fisheries management (Cresson *et al.* 2019, Lima *et al.* 2020).

As research into the impacts of artificial reefs has matured and reefs have been *in-situ* for greater periods of time, important longer-term data is becoming available to address the knowledge gap on how reef communities evolve. Bohnsack and Sutherland (1985) suggested community structure equilibrium is usually achieved within 1 to of 5 years. Conversely, studies at estuarine artificial reefs in Brazil suggest abundance, species richness, and biomass increase over a longer period of time (Lima *et al.* 2020). The rate of change in the fish assemblage at these Brazilian reefs was highest during the first six years after deployment (Zalmon *et al.* 2002, Santos *et al.* 2011, Santos and Zalmon 2015), but only stabilized fourteen years after reef deployment (Lima *et al.* 2020). Similar results have been reported by Santos *et al.* (2007) at the Algarve reef in Portugal, and over a ten year period at other sites by Relini *et al.* (2002) and Becker *et al.* (2022). The value of the artificial reef to local fisheries is therefore likely to change until the reef matures and stabilizes. This needs to be taken into consideration in any associated fisheries policy development and management planning.

A key consideration for fisheries management regarding habitat enhancement is the nature and extent of the effects on the abundance and biomass of exploitable species at a site. Unfortunately, few studies explicitly reported the impact of artificial reef installation on the exploitable biomass. As outlined above, many studies have also only been conducted over a relatively short time frame, and thus the fisheries outcomes reported may or may not be sustainable in the long-term. The overall trend observed has been for significant increases (at least in the short-term) in both the abundance and biomass of exploitable fish following the installation of artificial reefs (Jensen *et al.* 2000, Coutin 2001, Bortone *et al.* 2011, Vivier *et al.* 2021). Past studies have typically compared changes in fish biomass at artificial reef sites, before and after deployment, and to nearby natural reef references or bare substrate control sites. No study could be found that indicated the installation of an artificial reef decreased the fish biomass at a site. However, unrestricted fishing pressure has the potential to cause this if catch rates continue to exceed productivity in the target species. Several studies have reported artificial reefs to develop similar fish biomass to nearby natural reefs. However, most indicated biomass at artificial reefs to be between 2 to 26 times greater than bare substrate, with a median value of around a 5-fold increase (Table 2). Several studies also recorded biomass at artificial reefs to be between 2 to 5 times greater than nearby natural reefs.

Artificial reefs have also commonly been used to increase fish diversity, abundance and biomass in freshwater lakes, rivers and reservoirs. Many projects have been undertaken as part of environmental rehabilitation activities. However, there are also numerous examples where fisheries enhancement has been the primary project objective, whether through aggregation or increased productivity. In impoundments and reservoirs, installation of submerged structures has proven to be very effective at improving recreational angling by aggregating fish and is now a common strategy used by fisheries management agencies (Miranda 2016). In the USA, 80% of state fisheries agencies have undertaken habitat enhancement to improve their recreational lake and reservoir fisheries (Tugend *et al.* 2002). These habitat efforts have generally proven to be successful. In Lake Havasu, Arizona, fish abundance and angler catch and participation, more almost tripled following an extensive program of artificial reef installation (Jacobson and Koch 2008).

Table 2 Examples of the ecological impacts on exploitable species from the installation of artificial reefs or other habitat enhancement structures. Comparisons made with bare control areas, except where NR denotes comparison to a nearby natural reef.

Structure type	Source	Location	Abundance	Biomass	Growth rate	Recruitment
Concrete - coastal	Briones *et al.* 2007	Caribbean		7X		6X
Concrete - coastal	Ito 2011	Japan		6X	3X	increased
Concrete - coastal	Koeck *et al.* 2011	France	No change	No change		
Concrete - coastal	Lindberg *et al.* 2006	USA			1.15X	
Concrete - estuary	Folpp *et al.* 2013	Australia		7-20X 2X higher NR		
Concrete - estuary	Folpp *et al.* 2020	Australia	Adults - 10-75X NR Juveniles - 2-50X NR			
Concrete - estuary	Lima *et al.* 2020	Brazil	2X			
Concrete and steel	Kim *et al.* 2017	Korea		5X		
PVC	Rivlov & Benayahu 2002	Israel				2 orders of magnitude higher
Steel	Smith *et al.* 2016	Australia		5X NR		
Steel OAR	Champion *et al.* 2015	Australia				
Timber	Alam *et al.* 2020	Japan				
Vessels	Brock *et al.* 1994	Israel		26.5X		
Vessels	Sanchez-Caballero *et al.* 2021	Mexico	0.8X			

2.3.1.2 Productivity

Enhancement of fisheries through artificial structures may happen because of distributional changes of the existing biomass (*i.e.* attraction), or because of rising abundance (*i.e.* production). Understanding the relative value of artificial reefs acting as both fish attractors and fish producers has been one of the most controversial aspects of habitat enhancement. Production of new fish biomass at an artificial reef relies on the assumption that a latent biological productivity does not materialize because hard-substrate is a limiting factor (Broughton 2012, Roa-Ureta *et al.* 2019). The idea of equating fish production with the increase in the abundance of a stock per unit time is a difficult concept to demonstrate, and adequately evaluating the key drivers behind population dynamics in reef fish assemblages has proven to be exceedingly difficult. Many of these life history parameters may not be known and most populations fluctuate appreciably through time (Pondella *et al.* 2002). Additionally, fish may have pelagic egg and larval stages which complicates recruitment, because enhanced larval recruitment to habitat enhancement structures could be at the cost of nearby natural reefs (Szedlmayer and Shipp 1994, Lima *et al.* 2020). Movement of fish between different habitat types also confounds estimates (Smith *et al.* 2015).

Previous studies investigating these effects have supported both the attraction (Bohnsack 1989, Polovina 1989, Simon *et al.* 2011, Smith *et al.* 2015) and production (Szedlmayer and Shipp 1994, Brickhill *et al.* 2005, Cenci *et al.* 2011, Lima *et al.* 2020) hypotheses, indicating that both may occur under different conditions/situations and thus the results are context dependent on the design of the reef and the biological characteristics of the target species. To further confuse the situation, it appears that attraction and production both occur in the majority of artificial reef cases, described by Osenberg *et al.* (2002) as "… end points on a continuum" rather than discrete categories. It has been accepted by most that attraction is often detrimental to an ecosystem as it removes fish from natural reefs and can make them more accessible to be removed by commercial and recreational fishers. However, some studies have suggested that attraction may simply disperse fish biomass and make them harder for fisherman to catch (Smith *et al.* 2015).

Initially, artificial reefs were perceived as fish aggregation structures, which would increase fishing revenue (Santos and Monteiro 2007). In general, before and after comparisons in exploitable fish abundance or biomass between artificial reef sites and nearby bare control sites, indicate the impact of reef installation to be positive in the short-term (Jensen *et al.* 2000, Coutin 2001, Bortone *et al.* 2011, Vivier *et al.* 2021). However, simply attracting fish to a location only increases fishing efficiency and has no long term fisheries enhancement effect, due to local population loss (Grossman *et al.* 1997). In response to declining fisheries catches observed at some artificial reefs, the long-term value of artificial reef development has been questioned (Bohnsack 1989). Concern was raised over the ability of artificial reefs to locally aggregate the last few remaining fishes in a fishery, increasing their susceptibility to be caught, thus contributing to the decline and collapse of the resource.

It has been acknowledged that for any artificial reef to have any real and long-term fisheries enhancement capability, it must increase local productivity in order to augment natural fish production and thereby support local fisheries (Osenberg *et al.* 2002, Bull and Love 2019). For fisheries management, this issue is a critical component in the decision on how to use artificial reefs. The generation of new fish biomass increases fishery stocks without reducing their abundance in the surrounding areas, resulting in positive net regional benefits. Conversely, the attraction of biomass from surrounding areas re-distributes the existing biomass and whilst can increase the catchability at the artificial reef, might decrease the catchability in the surrounding areas (Pickering and Whitmarsh 1997). This is an extremely important question from a management perspective, because if the stocks are already identified as close to maximum sustainable yield, increased harvest without an associated increase in production will lead to population decline.

Increases in productivity can occur via the increased survival of juveniles, increased mating opportunities or indirectly by supporting increases in food availability due to epiphytic growth and other species inhabiting artificial habitats (Bohnsack *et al.* 1991, Pickering and Whitmarsh 1997, Molony and Bird 2005). The degree to which the increases in fish abundance at artificial reefs are caused by the production of new individuals, will depend on several factors, including the size of the artificial reef, its proximity to natural reefs, the strength of density-dependence, larval supply, and the extent to which the artificial reef and natural reef compete for larvae (Osenberg *et al.* 2002, Hackradt *et al.* 2011, Vivier *et al.* 2021).

Progress in the application of artificial reef for fisheries enhancement has been limited by doubts regarding the attraction versus production issue (Broughton 2012). Robust evidence that artificial structures can lead to productivity increases and thus long-term sustainable benefits in fisheries, will open the door to increased consideration of artificial reefs as a core tool of fisheries management strategies (Layman *et al.* 2016). There is now increasing evidence from rigorous scientific studies that artificial reefs which are well planned and implemented, do increase local productivity and enhance, rather than just attract fish populations (Johnson *et al.* 1994, Smith *et al.* 2015, Relini *et al.* 2002, Roa-Ureta *et al.* 2019, Becker *et al.* 2022).

It is generally agreed that artificial reefs provide habitat space and food resources for fishes (Peterson *et al.* 2003, Powers *et al.* 2003, Brickhill *et al.* 2005, Westerberg *et al.* 2013, Cresson *et al.* 2014, Champion *et al.* 2015). Bortone (2008) argued that artificial reefs may relieve pressure on populations on account of habitat bottlenecks. The provision of refuge in conjunction with greater access to food resources is most likely to drive any net increases in production of fish biomass on these reefs (Charbonnel *et al.* 2002, Powers *et al.* 2003, Fariñas-Franco and Roberts 2014). The cover provided by artificial structures has been demonstrated to reduce predation and increase biomass in a number of species (Herrnkind *et al.* 1997, Stephens *et al.* 2002, Lima *et al.* 2020). Additionally, growth of encrusting organisms and increased availability of zooplankton to grazing and zooplanktivorous fish, and amplified prey species abundance to higher order predators, can drive primary and secondary productivity at installed reefs (Lowry *et al.* 2010, Bortone *et al.* 2011, Champion *et al.* 2015, Becker *et al.* 2017, Folpp *et al.* 2020). Dietary studies (Leitão 2007, 2013) and isotopic analysis (Cresson *et al.* 2014) of fish have shown that artificial reefs increase secondary biomass production via trophic web interaction and consequently enhance fisheries.

Most artificial reefs are likely to initially experience high attraction, whilst productivity may take more time to develop as the reef ecosystem evolves, creating increasing levels of food and habitat resources (Relini *et al.* 2002, Zalmon *et al.* 2014, Cresson *et al.* 2019). If artificial reefs are enhancing production and the carrying capacity of reef habitat is limited, then appropriate reef installation should lead to increases in overall abundance, rather than just attracting fish from surrounding habitat. The rationale of the regional approach is that if fish are simply attracted to artificial reefs from the surrounding areas, then at the regional level there would be no increase in carrying capacity: the rise in abundance in the artificial reefs and its immediate vicinity would be cancelled by a proportional fall in abundance in the vacated surrounding region (Roa-Ureta *et al.* 2019).

Comparisons between fish populations at natural reefs and artificial reefs has provided evidence this occurs (Jensen *et al.* 2000, Coutin 2001, Bortone *et al.* 2011, Vivier *et al.* 2021). In estuaries, Folpp *et al.* (2020) found total fish abundance increased at artificial reef sites with no evidence of change at nearby natural rocky-reef sites. This provided evidence that the fish on artificial reefs were not attracted from the nearby rocky-reefs and were likely 'produced' by the addition of artificial reefs in these estuaries. Similar results were reported from long-term monitoring of an artificial reef complex in the plume of the Paraiba do Sol River in southeast Brazil. Over the long-term, the abundance of exploitable fish species was twice as high at the artificial reef compared to nearby control sites (Lima *et al.* 2020). Both sites exhibited increases over time, indicating production not just attraction.

Off Portugal, the carrying capacity of hard-structure in the Algarve region was increased through the installation of concrete artificial reefs, resulting in a 35% regional increase in exploitable sized *Diplodus sp.* (Roa-Ureta *et al.* 2019). The new production created by the deployment of artificial reefs spilled over in significant amounts to the surrounding areas. The increase was found to be precisely timed to occur after reef deployment, plus the delay due to the fish growing to commercial size (Gonclaves *et al.* 2003). Thus, the increase was unequivocally demonstrated to be linked to the reef installation. The significant rate of biomass increase was estimated at 226 tons per year and linked to both somatic growth and recruitment (Roa-Ureta *et al.* 2019).

Similar results have been reported at other offshore artificial reefs. A decade long monitoring program at the Loano artificial reef in Italy, found the reef to act in a similar manner to nearby natural reefs, with the new productivity helping to increase regional fish biomass (Relini *et al.* 2002). At a Californian artificial reef, DeMartini *et al.* (1994) used mark-recapture methods to show production at the reef occurred through both somatic and gonadal growth of the species present. Granneman and Steele (2014) demonstrated that for five economically important species, fish living on an artificial reef fared as well or better than those at nearby natural reefs for growth rate, body condition and reproduction. Total fish tissue production tended to be higher on artificial reefs than on natural reefs, though this pattern was not evident on all reef pairs. Tissue production was positively correlated with the abundance of large boulders, which was higher on artificial reefs than natural reefs. The similar or greater production of fish tissue per cubic metre on artificial reefs relative to natural reefs was used to conclude that the artificial habitats were valuable in producing fish biomass.

The behavioural tendency for tuna on other species to accumulate around floating objects (Fonteneau *et al.* 2000) has made the use of FADs extremely popular in purse seine fisheries and these devices are now widely distributed throughout the world's tropical and subtropical oceans (Bromhead *et al.* 2003). A significant proportion (one third) of the global purse sine catch are now taken using drifting FADs (Fonteneau *et al.* 2000). However, a large proportion of drifting FAD caught tuna comprises smaller size classes, consisting of Skipjack (*Katsuwonus pelamis*) and juvenile Yellowfin (*Thunnus albacares*) and Bigeye (*Thunnus obesus*) (Potier *et al.* 2001, Bromhead *et al.* 2003). These size classes were not exploited to the same extent prior to introduction of FADs. The intensive use of FADs to remove large numbers of small sized tuna has led to concerns that this fishing practice may substantially increase the risk of recruitment overfishing of these species and also contributed to the overharvest of some species (Marsac *et al.* 2000, IOTC 2002). Bycatch around FADs is also typically high (~10%) and there is some concern over the effects on local abundance of particular species such as turtles (Fonteneau *et al.* 2000). Concern has also been raised over the possibility that FADs may act as "ecological traps", luring tuna in unproductive regions and negatively impacting on their growth, condition and biological productivity (Marsac *et al.* 2000, Jaquemet *et al.* 2011, Hale and Swearer

2016). Together, these ecological concerns have resulted in a ban on the use of drifting FADs for purse seine fisheries in the Commonwealth waters of Australia (AFMA 2022)

In the South Pacific, communities have identified that FADs can be effective at aggregating pelagic species, particularly tuna, into areas more suitable for harvesting. The use of nearshore FADs can provide the opportunity for these communities to transfer some of their fishing effort away from the coral reefs to oceanic fisheries resources (Bell *et al.* 2015). This can help prevent over-exploitation of demersal coral reef fish species and help to maintain the normal representation of important functional groups of fish associated with reefs (Bellwood *et al.* 2004, Bell *et al.* 2011). Evidence suggests that up to 75% of fishing effort can be shifted using this approach (Sharp 2014, SPC 2012), as long as target pelagic stocks have a healthy population status. A similar concept has been trialled in Australia to see if nearshore FADs can reduce angler pressure on heavily harvested demersal sub-tropical species, such as Snapper (*Chrysophrys auratus*) and Pearl perch (*Glaucosoma scapulare*), by providing anglers with alternative high quality pelagic fishing options. The results of this work have yet to be published. However, Folpp and Lowry (2006) expressed some concern over the potential of angler catches at FADs to increase the harvest of juvenile fish, or of fish species considered to be fully exploited. This is a key consideration for the management of FADs. Of particular concern in more temperate areas is the impact of any additional angler effort directed at yellowtail kingfish, particularly juveniles in NSW.

Not all studies have demonstrated that artificial structures produce new fish. Bohnsack *et al.* (1994) compared fish counts on artificial reefs to two natural reef sites in south-eastern Florida, concluding that a significant proportion of the fish biomass was composed of adult fish attracted from natural reef sites. The review by Grossman *et al.* (1997) concluded that artificial reef production could have deleterious effects on reef fish populations, primarily through increased harvest pressure and access to previously unexploited fish stocks. They inferred that this was due to insufficient production occurring at the new habitat.

One of the core objectives of most habitat enhancement projects is to increase the biomass of exploitable species at a site. Changes in abundance and biomass have been recorded directly via a range of fishery independent techniques (*e.g.* Santos *et al.* 2007, Champion *et al.* 2015), or directly from trends in fishery catch data for exploitable biomass (*e.g.* Whitmarsh *et al.* 2008, Roa-Ureta *et al.* 2019), and are often used to infer a project’s economic success. When compared to control sites containing no structural habitat, increases in exploitable biomass or abundance typically range from 0 to 26 times, and 0 to 75 times, respectively (Table 2).

2.3.2 Socio-economic impacts

The published literature (both academic and professional) on fisheries habitat enhancement predominantly consisted of ecological studies, with only comparatively limited attention given to socio-economic evaluations. This corroborates the review by Becker *et al.* (2018) which found 233 of 270 studies on artificial reefs had a biological focus. Assessment of the socio-economic benefits accruing to communities provides insight into the degree to which the public benefit is being served by habitat enhancement and the economic consequences associated with enhanced habitat use for fisheries (Adams *et al.* 2011).

The economic worth of habitat enhancement projects is commonly assessed using metrics which describe either economic impact or economic value. Most studies only consider one of these measures. For example, they assess the economic impacts in terms of changes in industry output, employment, and other variables, whilst ignoring the welfare effects often measured in terms of consumer and producer surplus, compensating variation or non-use values (*e.g.* existence value). Consideration of both economic impact and value can provide more comprehensive understanding of the socio-economic benefits generated by a project because the two approaches represent different aspects of economic worth. However, as economic impact analysis does not take into consideration implementation cost, economic value is therefore more useful for comparisons between different projects because it allows for cost benefit comparisons (Whitmarsh and Pickering 2000).

A range of different methodologies have been used to analyse the socio-economic effects of habitat enhancement and descriptive terms have often been used interchangeably, but not always consistently or correctly. The degree of variation in methods, reporting details and the semantics of regional economic analysis confounds the comparability and interpretation of results (Milon 1991, Hudson 2001). Additionally, economic analysis concepts have not always been properly applied, resulting in erroneous analysis outputs. For example, Bell *et al.* (1989) mistakenly used estimated economic impacts rather than total economic value in their cost effectiveness comparison of several artificial reef designs. Where such methodological errors can be identified, the associated studies have been omitted from analyses in this review or only the relevant data included.

From a management perspective, marginal benefit or value-added is the most useful metric, because it identifies the change in value resulting from habitat enhancement and thus can be used to compare between different management options (Whitmarsh and Pickering 2000). At the simplest level, this involves comparison of the socioeconomic outcomes of an action to the base case scenario (*i.e.* status quo). In some cases, gross revenues have mistakenly been used in analyses rather than net benefits. Several studies reporting on the economic efficiency of artificial reef programs in Japan (*e.g.* Mottet 1981, Sato 1985) have used total revenues harvested from the reefs, rather than the change in the producer's profit (producer surplus). Unless the harvest prior to artificial reef construction was zero, the results presented were most likely significant overestimates of the actual economic benefits, since the change in profits forms only a small fraction of total revenue (Milon 1991).

In the current project, the literature review identified 26 primary studies that quantified the socio-economic value of habitat enhancement projects in sufficient detail to be included. Even fewer primary studies (7) were found which provided sufficient data on fisheries-related socio-economic impact assessment of habitat enhancement on a region (Table 3). Two additional reports summarised the results from multiple studies to determine the fisheries related socioeconomic values and impacts derived from habitat enhancement. The most economically well studied region was Florida in the USA, where more than half of studies have been conducted. For all other countries only one or two reports were identified from each, including Australia.

The socio-economic impacts of large-scale commercial applications of artificial habitat technologies have not been reported outside of Japan, Korea, Taiwan and China (*e.g.* Sheehy 1981, Sato 1985, Polovina and Sakai 1989, Kim *et al.* 2014). These studies have typically involved ranching, whereby artificial reefs were used to improve the survival and growth of wild and stocked fish for harvest. Artisanal fisheries were the primary target of enhancement projects in Europe and the Mediterranean Sea, but economic analyses could only be found from Portugal, Italy and Turkey. Similarly, in Brazil, Malaysia, Oman, Kenya, Costa Rica and India, only one or two economic studies were identified from each country. These studies again typically focussed on habitat enhancement for small-scale or artisanal fisheries, except for in Costa Rica where the economic value of using sunken vessels to enhance recreational tourism was evaluated. By comparison, a relative plethora of studies in the USA have investigated the direct benefits from offshore and nearshore artificial reefs installation for recreational activities. The literature review did not find any studies quantifying the socio-economic benefits of artificial reef installation in estuarine environments.

Table 3 The number of primary studies on fisheries habitat enhancement identified in the literature review from each country which used different economic assessment approaches

Country	Economic value	Cost-benefit analysis	Economic Impact
USA	13	8	6
Australia	2	2	-
Japan	2	2	-
Korea	2	2	1
Thailand	2	2	-
India	1	1	-
Taiwan	1	-	-
Turkey	1	-	-
Oman	1	-	-
Portugal	-	-	-
Argentina	1	-	-
Brazil	-	-	-
Italy	-	1	-
Philippines	-	1	-
Total	26	19	7

In Australia attention to the socio-economic aspects of habitat enhancement for recreational fisheries has been increasing. Recent studies undertaken in NSW and WA have provided comprehensive information on the socio-economics of offshore artificial reefs and other man-made structures (Nous 2019, Harvey *et al.* 2021). A further study is currently underway in Tasmania looking at the direct and indirect benefits of two artificial reefs and five FADs (FRDC project 2020-073, University of Tasmania). The information from these studies will be extremely valuable in guiding future habitat enhancement investment and projects in Australia. Only limited occurrences of habitat enhancement for commercial fisheries applications have occurred. These projects have focussed on commercial ranching and thus much of the data associated with such projects is commercial-in-confidence (*e.g.* Ocean Grown Abalone's artificial reefs in Western Australia).

All economic values presented in this section have been converted to 2021 Australian dollars unless otherwise indicated, to enable easier comparison between results.

2.3.2.1 Economic Impact

Some valuation studies on habitat enhancement include economic impact assessments to quantify the increased economic activity that a project brings to a region. Economic impact assessments quantify the gross changes in a region's existing economy that can be attributed to a given industry, event, or policy, such as a habitat enhancement project (Watson *et al.* 2007).The impacts are usually described in terms of the total expenditures (economic output), number of jobs, local income, tax revenue and the value-added generated by the habitat enhancement project. For recreational benefits (*i.e.* angling), significant economic impacts are only realized if the recreation opportunities created are sufficiently valued to attract additional visitors to the region or increase the expenditures of visitors already there (Milon 1989, Whitmarsh *et al.* 2008).

The scale of the economic impacts on local economies from installing artificial reefs can vary considerably (Sutton and Bushnell 2007). In general, studies have found the economic impacts of habitat enhancement projects to be very positive (Table 4), particularly for recreational fisheries. The size of the impact is closely related to scale of the habitat enhancement activity, but there is also a trend for declining marginal benefits with scale (Sutton and Bushnell 2007). Reported economic impacts included in the literature review are summarised in Table 4.

Fisheries related economic impacts from habitat enhancement projects are most well studied in the USA. Florida has one of the world's most extensive systems of artificial reefs, with approximately 3,750 reefs deployed across 34 coastal counties (Ropicki *et al.* 2021). The artificial reefs are well utilized by recreational anglers, divers, and other user groups (Adams *et al.* 2011), and their existence has a significant economic impact. Several detailed economic impact assessments have been undertaken in an attempt to measure these economic benefits for nearby coastal communities. Huth (2014) estimated the economic impact derived from recreational fishing use of artificial reefs for the whole of Florida state, to include $3,193 million in economic output, $1,307 million income, 25,821 jobs and $261 million in tax revenue for the state. The impacts from fishing activities were almost double that for diving related activities on the artificial reefs.

At a more local scale, Johns *et al.* (2001) calculated that the total economic impact of artificial reefs in four Florida counties. The largest estimated economic impact occurred in Broward County, where $2,975 million in sales, generated $1,552 million in income, and provided 16,800 jobs. For the three other southeast Florida counties, the economic impacts ranged from $391 million to $1,296 million in sales, $101 million to $603 million in income, and 1,800 to 6,000 jobs per county (Table 4). Of these values, approximately half was related to fishing activity, whilst the remained were associated with snorkelling, scuba diving and site-seeing. Similarly, Johns (2004) estimated the annual economic impact derived from artificial reefs in Martin County to be $14.1 million in expenditures, $6.3 million in income and 99 jobs, with approximately 86% of this attributable to recreational fishing. Swett *et al.* (2011) estimated recreational use of artificial reefs in six counties of southwest Florida had substantially lower economic impacts, with a combined annual economic output of $295.1 million. Recreational angling contributed approximately 67% of the impact, generating an economic output of $177.1 million and a total value added impact of $107.9 million per year. At the county scale, the annual economic impacts from fishing ranged from $15.2-59.2 million in expenditures, $8.3-30.9 million in income, 157-575 jobs, $1.2-4.2 million in taxes and $9.4-35.1 million value added (Table 4). In northwest, Florida, Bell *et al.* (1998) reported economic impacts of $414 million in expenditures, 8,136 jobs and $84 million in income derived from recreational use of artificial reefs by anglers, divers and snorkellers.

In Korea, establishing sea fishing ranches using artificial reefs in the Gyeong-Nam Province delivered significant economic impacts to the region. Construction of the Large Sea Ranch increased commercial fish production by more than five-fold and increased fishermen's incomes by 26% (Kim *et al.* 2017). Additionally, recreational anglers spent $26 million at the site (Pyo 2009). Using a multi-regional impact model, Kim *et al.* (2017) calculated a newer ranch, the Small Sea Ranch, had an annual impact of $39.1 million total output, 677 jobs, $13.6 million income, $0.3 million tax revenue and $8.7 million value-added. Of these figures, approximately 70% accrued within the province, with the remainder spread amongst other regions.

Only one economic impact assessment on habitat enhancement in freshwater fisheries was identified in the literature review. The freshwater fisheries enhancement programs undertaken in Lake Havasu, Arizona, was one of the largest (cost $86.6 million over 10 years) enhancement projects undertaken to rectify extensive declines in recreational fishery quality. Significant degradation of fish habitat due to reservoir aging was identified as one of the primary causes (Jacobson and Koch 2008), so the program focussed on improving structural complexity within the reservoir through the installation of artificial reefs made from natural and synthetic materials. The program was so successful at improving the fishery, that annual angler use increased from 43,000 angler days to 170,000 angler days. The resulting increase in the associated fishing related expenditures, have produced significant, long term socioeconomic benefits to the local area, including increases in employment, income and tax revenues. Anderson (2001) estimated annual non-resident fishing expenditure value-added $56.0 million, generated 655 jobs, $34.9 million in income, $17.6 million in tax revenues and stimulated an increase of $104.5 million in expenditure per year. In addition, resident anglers' expenditures in the local area generated an additional $54.7 million in value-added , 639 jobs, $17.6 million in tax revenues and an increase of $106.7 million in expenditure per year. These benefits were expected to last into the foreseeable future with relatively low ongoing program and structure maintenance costs.

The literature review found no economic impact assessments conducted in Australia on fisheries habitat enhancement projects. The identified economic studies all reported on economic values instead (see below).

Table 4 Literature review summary of fisheries related economic impacts from habitat enhancement projects.

Study	Country	Location	Enhancement	Expenditure ($AUD million)	Jobs	Income ($AUD million)	Taxes ($AUD million)	Value-added ($AUD million)
Bell *et al.* 1998	USA	NW Florida	Coastal and offshore AR	1,068.5	8,136	216.8	-	-
Johns *et al.* 2001[1]	USA	Palm Beach County	Coastal and offshore AR	458.2	1,756	161.8	-	-
Johns *et al.* 2001[1]	USA	Broward County	Coastal and offshore AR	2,975.3	16,821	1,552.0	-	-
Johns *et al.* 2001[1]	USA	Miami-Dade County	Coastal and offshore AR	1,296.2	5,990	603.7	-	-
Johns *et al.* 2001[1]	USA	Monroe County	Coastal and offshore AR	391.1	2,319	100.5	-	-
Johns 2004	USA	Martin County	Coastal and offshore AR	12.1	85	5.4	0.78	-
Anderson 2001	USA	Arizona	Freshwater AR	211.1	1,294	95.5	35.10	110.7
Swett *et al.* 2011[2]	USA	Manatee County	Coastal and offshore AR	15.2	157	8.3	1.16	9.4
Swett *et al.* 2011[2]	USA	Sarasota County	Coastal and offshore AR	23.6	226	13.0	17.64	14.7
Swett *et al.* 2011[2]	USA	Hillsborough County	Coastal and offshore AR	21.0	190	11.4	1.52	12.9
Swett *et al.* 2011[2]	USA	Pinnellas County	Coastal and offshore AR	59.2	575	30.9	4.17	35.1

Study	Country	Location	Enhancement	Expenditure ($AUD million)	Jobs	Income ($AUD million)	Taxes ($AUD million)	Value-added ($AUD million)
Swett *et al.* 2011[2]	USA	Charlotte County	Coastal and offshore AR	17.7	205	9.2	1.30	10.5
Swett *et al.* 2011[2]	USA	Lee County	Coastal and offshore AR	40.4	385	22.2	3.03	25.3
Swett *et al.* 2011[2]	USA	SW Florida	Coastal and offshore AR	177.0	1739	95.0	129.53	107.9
Huth 2014	USA	Florida	Coastal and offshore AR	3,192.9	25,182	1,306.6	261.42	-
Kim *et al.* 2017	Korea	Gyeong-Nam Province	AR ranch	39.1	554	13.6	0.28	8.7

[1,2] The full economic impacts reported in Johns *et al.* (2004) [1] and Swett (*et al.* (2011) [2] were associated with recreational fishing, snorkelling and diving use. Approximately 86% and 67% respectively, were attributable to recreational fishing, therefore only these portions have been included in the table.

2.3.2.2 Economic value

The systematic review found 26 studies that quantified the economic value that fishery enhancement projects provided to stakeholders. The economic value was identified by measuring the value that both users and non-users placed on the opportunity and experience of using habitat enhancement resources such as artificial reefs or FADs. The extent to which users value these resources was expressed by the money they spend to use them, plus any additional amount they were willing to pay before foregoing the opportunity to use that resource (Huppert 1983). Far fewer studies (5) presented information on marginal benefits from consumer and producer surplus, limiting the number of studies where cost benefit evaluations were undertaken.

Both users and non-users were found to derive value from habitat enhancement projects, although most indirect and existence values were rarely reported. Non-users were clearly shown to be willing to pay for habitat enhancement in some scenarios, even when they have not used such resources in the past (Bockstael *et al.* 1986, Milon 1989, Whitmarsh and Pickering 2000). In the current review, the scope for the discussion on economic value was restricted to the value provided to fisheries from habitat enhancement. Significant value can also be derived from other non-extractive activities, such as snorkelling or scuba diving, but these are not the primary objective from enhancement activities that target fisheries enhancement (Milon 1988, Brock *et al.* 1994, Chen *et al.* 2013, Huth 2015, Harvey *et al.* 2021).

The studies reporting on the economic value of habitat enhancement projects had a reasonable global distribution (10 countries), with the USA again the most well studied region (Table 5). The included studies encompassed habitat enhancement using a variety of artificial reef and FAD types, and deployment purposes. The most commonly reported scenarios related to purpose-built concrete coastal artificial reefs for recreational or artisanal fisheries. Unfortunately, the studies covered such a broad range of habitat enhancement approaches (*e.g.* concrete reefs, vessels, materials of opportunity, mixed structures) that it was not possible to clearly determine the influence on value of the specific features from different habitat types. However, where possible similar scenarios have been grouped to assist comparisons of the benefits achieved.

Only two studies from Australia were identified by the literature review. Both studies addressed the value of purpose-built coastal/offshore artificial reefs to recreational anglers, which appears to be substantial. A report on the social return on investment for two of the offshore artificial reefs in the NSW Artificial Reef Program for recreational fishers found that investment in the reefs generated a positive net return for NSW, provided high quality fishing opportunities, generated new research activity and encouraged increased recreational activity that delivers flow on economic and social impacts for communities (Nous 2019). Direct use of the reefs is limited to recreational fishers only, thus producer surplus and non-extractive user values were considered negligible and thus not reported. At the Port Macquarie offshore artificial reef, the value derived by fishers from reef use was estimated over the 30-year lifespan of the reef to be in the order of $57.8 million. Similarly, the value to fishers of the Southern Sydney “JD” artificial reef over the 30-year lifespan of the reef was estimated to be in the order of $58.5 million. Contingent valuation methods estimated anglers using these reefs were willing to pay an additional $26.35 per annum, whilst non-users were willing to pay $15.81 per annum.

In Western Australia, Harvey *et al.* (2021) estimated the installation of the Exmouth Integrated Artificial Reef would generate additional trips to the region and/or improved fishing quality. If the reef leads to substitution of fishing sites, the annual marginal benefit from reef installation would be $114,500. Alternatively, if the new reef results in a significant increase in the number of fishing trips to the region, the annual consumer surplus would increase by $267,000. These values were considered conservative, because they only included limited information about any additional benefits to divers, charter boat operators, commercial fisheries and no estimates on the willingness to pay for non-user and existence values.

Recreational use of habitat enhancement sites has been reported to generate substantial economic value around the world. The most well studied region was Florida in the USA, where more than 3,750 artificial reefs have been deployed (Ropicki *et al.* 2021). One of the earliest studies was conducted in Miami-Dade County, where Milon (1988) valued the annual benefits to residents for developing new artificial reefs at between $135,500 and $3,161,600. The broad spectrum of values resulted from comparison of eight different valuation approaches, considering both users and non-users. The lower bounds were provided by five different travel cost methods focussing on recreational anglers only, with annual values of $135,500 - $456,000. The travel cost models indicated that site use was positively related to gross catch rates and negatively related to travel costs. Conversely, contingent valuation methods provided values of $543,800 to $3,161,600. Extrapolating these results to the Miami-Dade County artificial reef system existing at the time (1985), generated a present value (3% discount) with a lower bound from the travel cost approach of $78.0 million and an upper bound from the contingent valuation approach of $571.9 million. These values would be an underestimate, as only private boat users, but not charter operators or commercial fishers were included.

Johns (2004) measured the economic values associated with recreational fishing at the nearby Martin County artificial reef and found the total yearly utilization value of the existing artificial reefs was $7.12 million. This was higher than the Milon (1987) estimates due to demographic differences. Based on user and non-user willingness to pay, the annual value for new artificial reefs was also significant at $2.21 million. Johns *et al.* (2001) estimated similar annual use values for artificial reef systems in Southeast Florida, which ranged from $4.7 million to $29.0 million per county, with total economic values between $520 million to $2,184 million (Table 5).

Artificial reefs overseas have also been found to generate direct use values from commercial fishing (Brock *et al.* 1994, Kasim *et al.* 2003, Vivekanandan *et al.* 2006, Whitmarsh *et al.* 2008, Islam *et al.* 2014). For example, in Chumphon Province, Thailand, the annual fishery income for a small-scale community fishery increased by 6% 3 years after an artificial reef deployment program (Kantavichai *et al.* 2019). However, a mixed response was observed, with 24% of households experiencing decreased fishery income, 38% experiencing no change, and 38% experiencing an increase. Despite the mixed results, most of the respondents showed satisfaction toward artificial reef program in enhancement of marine resources and environment (Kantavichai *et al.* 2019).

A total of 17 studies were found which compared fisheries economic values between artificial reefs and naturally occurring reefs. Bockstael *et al.* (1986) concluded that anglers were only more likely to use the artificial reefs if catch rates were higher and travel costs lower than the natural habitat alternatives. In general, the value of artificial reefs to local fisheries was typically equal to or higher than nearby natural reefs, but the relative scale of reef availability needs to be taken into consideration. In almost all areas, the availability of artificial reefs is far lower than that of naturally occurring reef systems, even in areas with the highest density of artificial reef deployment (Ropicki *et*

Table 5 Literature review summary of fisheries related economic values for habitat enhancement projects.

Study	Country	Enhancement type	Value type	Data collection year	Valuation method	Valuation context	Value (study units)	Value (AUD 2021)
AlOufi *et al.* 2018	Oman	Concrete and steel AR	Use - Direct (com)	2018	Market	Rise in revenue associated with commercial fishing	2,400,000 OR per year per fishery	$9,225,264 per year per fishery
Bell *et al.* 1998	USA	Mixed AR	Use - Direct (rec)	1997-1998	Market	Revenue from user expenditure related to AR	19,700,000 USD per year	$50,844,963 per year
Brandini *et al.* 2014	Brazil	Concrete AR	Use - Direct (rec)	1998-2003	Market	Revenues from rec fishing in study period	70,933 USD per year	$144,491 per year
Buchanan 1973	USA	Tyres and vessels AR	Use- Direct (rec)	1972	Market	Total expenditure associated with AR	36,000 USD per 4 month season	$235,086 per 4 month season
Choi 2013	Korea	Concrete	Use - Direct (com)	2013	Market	Net income from commercial fishing	1,167,680 Won per vessel per year	$1,518 per vessel per year
Crabbe & McClanahan 2006	Kenya	Vessels	Use - Direct (com)	2004	Market	Revenue associated with commercial fishing	9.00 USD p.p. increase per day	$17.69 p.p. increase per day
Islam *et al.* 2014	Malaysia	Mixed AR	Use - Direct (com)	2011	Market	Revenue associated with commercial fishing at AR	$164 USD less p.p. per month	$191.00 less p.p. per month
Kasim *et al.* 2003	India	Concrete AR	Use - Direct (com)	2007	Market	Net income from commercial gillnet fishing	1252 INR per unit operation per year	$37.66 per unit operation per year

Study	Country	Enhancement type	Value type	Data collection year	Valuation method	Valuation context	Value (study units)	Value (AUD 2021)
Kasim *et al.* 2003	India	Concrete AR	Use - Direct (com)	2007	Market	Net income from commercial hook and line fishing	4650 INR per unit operation per year	$168.79 per unit operation per year
Kim *et al.* 2017	Korea	Concrete and steel AR	Use - direct (rec)	2015	Non-market (TC)	Total additional revenue associated with recreational fishing	32,400,000,000 Won per ranch per year	$39,055,000 per ranch per year
Ramos *et al.* 2006	Portugal	Concrete AR	Use - Direct (com)	2002	Market	Net income associated with artisanal fishing at AR	€7858 to €18896 per fisherman per year	$12,388-29,790 p.p. per year
Vivekanandan *et al.* 2009	India	Mixed AR	Use - Direct (com)	2003	Market	Income associated with artisanal fishing	71.3 RS extra per hour of operation	$4.03 extra per hour of operation
Whitmarsh *et al.* 2008	Portugal	Concrete AR	Use - Direct (com)	2005	Market	Value per unit effort associated with artisanal fishing	13 € extra per unit effort on AR	$19.64 extra per unit effort on AR

al. 2021). Johns *et al.* (2001) also noted that there was likely to be a declining marginal value with increasing number of reefs (artificial or natural) in an area. However, artificial reefs have generally occurred against a backdrop of degraded natural systems, or in areas where natural reef habitat was limited (Harvey *et al.* 2021), so the initial marginal decline is expected to be low in most circumstances before saturation is reached (Bell *et al.* 2003).

In general, artificial reefs provided an equal or higher economic value to local small-scale fisheries, when compared to nearby natural reefs. At the Algarve reef system in Portugal, Whitmarsh *et al.* (2008) estimated artisanal fishers at the artificial reefs initially earned $33.82 per unit effort more than at non-enhanced sites, with this value increasing monthly by $0.46 as the reef system matured. It was concluded that fishers using the Algarve reef system would earn 1.73 times what they would if they fished elsewhere.

The deployment of artificial reefs has been reported to improve fishing income, despite the fishing yield sometimes being the equal to or lower than yield at natural reefs. The increased income results from a greater proportion of high value species in the catch at artificial reefs. In Tamil Nadu, India, the average monthly income of commercial fishermen was almost three times higher when fishing artificial reefs compared to un-enhanced areas, regardless of the gear type the used (Kasim *et al.* 2013). Catch rates were similar between the areas, but higher catch value per unit effort from artificial reefs led to higher income. Similarly, in Chennai, Vivekanandan *et al.* (2006) found catch per unit effort to be almost three times lower at the artificial reefs than nearby areas. However, the value of the catch was significantly higher from artificial reefs, resulting in 36% higher income per hour. The difference in catch per unit effort was attributed to the different gear used between artificial reef and non-artificial reef sites.

The use of artificial reefs for habitat enhancement has not always been shown to be economically beneficial to local fisheries. Islam *et al.* (2014) reported that producer income for artisanal fishers using artificial reefs in the Terrengganu Peninsula, Malaysia, decreased after reef installation, but this was dependent upon the gear type they used. Overall, average monthly income declined by 23%, compared to fishers who used nearby natural reefs. Benefits were only observed in the hook and line fishery, which could fish in very close proximity to the reef structures. For these fishers, average monthly income increased by 13% following reefs installation. Conversely, net and trap fishermen who could not fish close to the reefs experienced significant declines in income. Crabbe and McClanahan (2006) similarly found little benefit for local commercial fishers from the scuttling of two large vessels in Kenya. Before and after comparisons at the wrecks and nearby sites showed a short-term increase in fish catch for speargun fishermen but a decrease for hook and line fishermen and no evidence for the sustainable use. The economic benefit to the speargun fishermen was estimated to only be an additional $1,966 annually, although from the use of semi-structured interviews, the fishermen perceived little benefit from the shipwrecks. In contrast, the economic benefit from enhanced dive tourism was estimated as $147,500-$342,000 annually and there was generally a high awareness by all stakeholders of the social benefits of the shipwrecks to the local community.

The recreational fishing value for artificial reefs is often lower than that of natural reefs. In southeast Florida, Johns *et al.* (2001) estimated the value recreational users placed on reefs across four counties by determining the maximum amount of money that reef users were willing to pay to maintain both natural and artificial reefs in their existing condition, and to add more artificial reefs to the system. Natural reefs were more highly valued than the artificial reefs. Contingent valuation revealed total annual use values for all users of $166 million for maintaining the existing artificial reefs, $500 million for maintaining natural reefs, and $52 million per year for the establishment of new artificial reefs. The relative capitalised values (3%) for these were estimated at $5,500 million, $14,900 million, and $1,746 million, respectively. Approximately half of these values were attributed to recreational fishing, with the remainder from recreational diving, snorkelling and other non-extractive reef activities. Johns (2004) reported a similar, but less pronounced, trend in use values between artificial and natural reefs in Martin County, Florida. The estimated total use value of artificial reefs was $27.68 per person per day, equating to an annual use value for all users of $7.12 million. In comparison, the values for natural reefs at the same location were $30.43 per person per day and $7.82 million annually. The capitalised (3%) use values were $236 million and $261 million, respectively.

The marginal value of habitat enhancement projects is highly dependent upon the number of stakeholders for that site or region. The presence of more stakeholders typically results in greater total economic value estimates due to aggregative effects. This makes it somewhat difficult to directly compare the values generated at these sites to compare effectiveness in different fishery scenarios. Metrics quantifying benefits per standardised unit are more suited for direct comparisons and enable estimation of the potential value of new projects (‘benefit transfer’). For commercial fisheries, defining benefits by the change in value per unit effort (VPUE) or profit per vessel per unit time provide good metrics for comparisons to be made on. For non-market values, including recreational fisheries, stakeholder willingness to pay provides an excellent metric that facilitates comparison of outcomes between projects and also enables total value to be calculated.

The literature review identified 12 studies providing sufficient information on non-market willingness to pay values for previous habitat enhancement projects, primarily using the travel cost and contingent valuation methods(Table 6).

Willingness to pay for maintaining or installing new artificial reefs was relatively consistent across studies (Table 6). Overall, commercial fishers were willing to pay more than recreational fishers, most likely reflecting the direct economic benefit they receive through increased harvest value or profit. Non-market valuations found recreational angler willingness to pay for existing artificial reefs averaged $12.62 per day (range $7.97-20.54), $16.44 per trip (range $10.57-27.67) or $14.00 per year (through license fee increases). By comparison, only two studies reported on the willingness of commercial fishers to pay for maintaining existing habitat enhancements, estimating a daily value of $17.69 per person for artificial reefs or annual value of $147.11-169.39 per person for FADs (Table 6).

In one of the few studies found on the economic value of FADs, Samples (1986) used three contingent valuation methods with recreational and commercial fishers using the FAD system in Hawaii to estimate the value of the FAD program. When asked to nominate an annual amount a fisher was willing to donate to continue the FAD program, willingness to pay for recreational anglers was $94, part-time commercial fishers $147 and full-time commercial fishers $169. This provided an average across all samples of $129 per year. Alternative approaches using random dichotomous voting for a one-off payment to keep the FAD program going for one year, resulted in combined willingness to pay of either $254 or $397, depending upon how the statistics were undertaken. Using the mean of the three methods ($249.88), the FAD program was estimated to have a value of $824,300. This value was likely to be a significant underestimate since two key FAD user groups were not accounted for: passengers on recreational vessels ($2,050,500 calculated from information in

Samples 1986) and the pole and line tuna fishers ($313,000, Sproul 1984). If the benefits for these two groups are included, the annual economic value rises to around $3,188,000.

Willingness to pay for the installation of new artificial reefs was found to be similar to that for maintaining reefs, although the per trip and annual amounts were higher, and the ranges were slightly larger. For new reefs, recreational anglers were willing to pay an average of $11.11 per day (range $4.27-26.69), $39.19.44 per trip (range $38.13-40.25) or $34.35 per year (range $15.81-220.00) (Table 6). No studies were identified in the literature review reporting on the willingness of commercial fishers to pay for new artificial reefs.

Very little difference between the willingness to pay for habitat enhancement was reported between visitors and residents (Table 6). The review results also suggested recreational fishers who used artificial reefs and FADs typically had higher willingness to pay than non-users. This trend was mirrored in willingness to pay for the development of new structures. For example, McGurrin and Fedler (1989) found artificial reef users were willing to pay almost twice as much as recreational fishers who did not utilise those reefs. Similarly, Bockstael *et al.* (1986) found users were willing to pay one and a half times as much to support continuation of the artificial reef program in South Carolina, USA. Although the willingness to pay for non-users was lower, the research indicated that they receive some benefits from habitat enhancement due to expected future use, indirect values (such as stock enhancement, congestion) or extrinsic existence values (Whitmarsh and Pickering 2000). Whilst the values were only relatively small per person, the number of non-users is typically higher and their aggregated benefits, can add up to a substantial amount.

Table 6 Literature review summary of willingness to pay for fisheries habitat enhancement projects. CVM = contingent valuation method, DCE = discrete choice experiment, MV = market value, and TC = travel cost.

Study	Country	Enhancement type	Value type	Data collection year	Valuation method	Valuation context	WTP (study units)	WTP (AUD 2021)
Chen *et al.* 2013	Taiwan	Concrete AR	Use - Direct (rec)	2008	Non market (TC)	Travel costs associated with rec fishing	281.89 USD per trip	$570.55 per trip
Chen *et al.* 2013	Taiwan	Concrete AR	Use - Direct (rec)	2008	Non market (CVM)	WTP for new AR for rec fishing	13.00 USD per trip	$26.31 per trip
Harvey *et al.* 2021	Australia	Concrete and steel AR	Use - Direct (rec)	2021	Non market (CVM)	WTP for AR for rec fishing	178 AUD p.p. per year	$178 p.p. per year
Huth 2016	USA	Mixed AR	Use - Direct (rec)	2016	Non market (CVM)	WTP licence increase for new AR for rec fishing	resident - 32.47 visitor - 31.78 USD per year	resident - $49.37 visitor - $48.32 per year
Huth 2016	USA	Mixed AR	Use - Direct (rec)	2016	Non market (CVM)	WTP for charter clients for new AR for rec fishing	resident - 26.47 visitor - 25.08 USD per trip	resident - $40.25 visitor - $38.13 per trip
Johns *et al.* 2001	USA	Mixed AR	Use - Direct (rec)	2000	Market (MV) and non market (CVM)	WTP extra in boat registration and charter costs for new ARs for all SE Florida	licence - 8.63 USD p.p. per day or charter fees -75 USD p.p. per year	licence - $26.69 p.p. per day or charter fees - $220 p.p. per year
Johns *et al.* 2001	USA	Mixed AR	Use - Direct (rec)	2000	Market (MV) and non market (CVM)	WTP extra per trip to maintain existing AR for all SE Florida	2.72 USD p.p. per day	$7.97 p.p. per day

Study	Country	Enhancement type	Value type	Data collection year	Valuation method	Valuation context	WTP (study units)	WTP (AUD 2021)
Johns *et al.* 2001	USA	Mixed AR	Use - Direct (rec)	2000	Nom market (DCE)	WTP for new AR Palm Beach County	6.47 USD p.p. per day	$10.42 p.p. per day
Johns *et al.* 2001	USA	Mixed AR	Use - Direct (rec)	2000	Nom market (DCE)	WTP for new AR Broward County	14.07 USD p.p. per day	$12.22 p.p. per day
Johns *et al.* 2001	USA	Mixed AR	Use - Direct (rec)	2000	Nom market (DCE)	WTP for new AR Miami-Dade County	3.5 USD p.p. per day	$4.27 p.p. per day
Johns *et al.* 2001	USA	Mixed AR	Use - Direct (rec)	2000	Nom market (DCE)	WTP for new AR Monroe County	6.36 USD p.p. per day	$4.52 p.p. per day
Johns 2004	USA	Mixed AR	Use- Direct (rec)	2003	Non market (TC)	WTP for current AR Martin County	14.08 USD per trip	$27.67 per trip
Johns 2004	USA	Mixed AR	Use- Direct (rec)	2003	Non market (CVM)	WTP to pay for a new AR Martin County	4.33 USD p.p. per day	$8.51 p.p. per day
McGurrin & Fedler 1989	USA	O&G Rigs	Use - Direct (rec)	1989	Non market (CVM)	WTP for a new AR	user – 19.38 nonuser -10.00 combined -14.36 USD p.p. per reef	user – $55.22 nonuser - $28.29 combined -$40.92 p.p. per reef
Milon 1988	USA	Vessels	Use - Direct (rec)	1985	Non market (CVM)	WTP to pay for a new AR	user - 18.04-26.57 nonuser - 1.14-31.93 USD p.p. per year	user - $80.42-118.44 nonuser - $5.08-142.34 p.p. per year

Study	Country	Enhancement type	Value type	Data collection year	Valuation method	Valuation context	WTP (study units)	WTP (AUD 2021)
Milon 1988	USA	Vessels	Use - Direct (rec)	1985	Non market (TC)	WTP for current AR	3.14 USD p.p. per year	$14.00 p.p. per year
Morgan *et al.* 2018	USA	Vessels	Use - Direct (rec)	2014	Non market (CVM)	WTP for increased recreational license fee	resident – 32.60 visitor - 33.33 USD p.p. per year	resident – $41.85 visitor – $42.79 p.p. per year
Nous 2019	Australia	Concrete and steel AR	Use - Direct (rec)	2018	Non market (CVM)	Annual WTP for new AR for rec fishing	user - 25.00 non-user - 15.00 AUD p.p. per year	user – $26.35 non-user - $15.81 p.p. per year
Samples 1986	USA	FAD	Use - Direct (com)	1984	Non market (CVM)	WTP to maintain FADs for commercial fishing	33-38 USD p.p. per year	$147.11-169.39 p.p. per year
Samples 1986	USA	FAD	Use - Direct (rec)	1984	Non market (CVM)	WTP to maintain FADs for rec fishing	21 USD p.p. per year	$93.61 p.p. per year
Samples 1986	USA	FAD	Use - Direct (rec)	1984	Non market (CVM) 3 x different approaches	Mean WTP to maintain FADs for both rec and commercial fishing	29-89 USD p.p. per year	$124.82-395.74 p.p. per year
Tunca *et al.* 2016	Turkey	Concrete AR	Use - Direct (rec)	2016	Non-market (CVM)	Recreational user WTP for AR	Resident - €6.82 Visitor -€7.15 p.p. per trip	Resident - $10.57 Visitor - $11.09 p.p. per trip
Tunca *et al.* 2016	Turkey	Concrete AR	Use - Direct (com)	2016	Non-market (CVM)	Commercial user WTP for new AR	€7.36 p.p. per trip	$11.41 p.p. per trip

2.3.2.3 Cost benefit analysis

There is a strong need for quantitative evaluations of the efficiency of habitat enhancement projects to ensure they deliver appropriate benefits and optimise outcomes from public investment. Despite this need, the number of studies undertaking cost-benefit analyses has been limited. The literature review only identified 12 studies which included cost benefit analysis on the value generated by habitat enhancement. All of the studies were for coastal and offshore structures. No analyses were identified for habitat enhancement in estuarine or freshwater environments. Of these, two used absolute value rather than marginal benefit (Sato 1985, Bell *et al.* 1998), two only reported payback periods, and one study only provided the NPV. Details on the other seven studies have been summarised in Table 7. Outside of Florida, the BCR for artificial reefs were all positive, but typically not large, ranging from 1.08-1.29, with mean of 1.46. The BCR values reported from Florida were substantially higher, but more variable, ranging from 5.8-23.3. Only one study was identified where cost benefit analysis had been conducted on FADs. The results were positive, but there is some uncertainty over the scale of the value (see below).

Only a single cost benefit study on artificial reefs was identified from Australia. Nous (2019) undertook cost benefit analyses on two offshore artificial reefs in NSW. It was estimated that the Port Macquarie artificial reef involved investment of approximately $1.1 million in the reef design, construction and deployment and annual research and administration costs for the reef were approximately $101,200 per annum. The socio-economic value (NPV) added by the Port Macquarie artificial reef was estimated to be $200,000 (7% discount rate, 30 years). The estimated value to fishers of all fishing use over the lifespan of the reef was $57.8 million. This equated to an IRR of 8.8% and a BCR of 1.18 per construction dollar. That is, approximately $0.18 of value was created for every $1.00 invested in the construction of the reef, over and above the 7% real return on the funds invested. Including maintenance costs over the reef's projected life of 30 years, the BCR drops to 1.01. At the southern Sydney "JD" Artificial Reef, initial costs were estimated to be $2.3 million, with ongoing costs of $104,600 per annum. The estimated value to fishers of all fishing use over the lifespan of the reef was $58.5 million. The NPV added by the Southern Sydney "JD" artificial reef was estimated to be $223,000 (7% discount rate, 30 years), equating to an IRR of 9.0% and a BCR of 1.10 per construction dollar. Including maintenance costs over the reef's projected life of 30 years, the BCR also drops to 1.01.

No examples of cost benefit analysis for habitat enhancement in freshwater fisheries was identified. However, existing information can provide insight into the likelihood of this approach delivering positive net benefits to inland fisheries. Recent research in Queensland investigating the use of artificial reefs and FADs in stocked impoundments has found that angler catch rates and satisfaction with their fishing experience improved following reef installation (Norris *et al.* 2021). Freshwater artificial reefs usually have a low cost to create and typically consist of multiple smaller clusters of individual units. Norris *et al.* (2021) reported the cost per unit ranged from $6.90 up to $612 depending upon the structure type. Therefore, construction and deployment of these structures into impoundments at a sufficient scale to benefit recreational angler experience is likely to cost $20,000-$100,000 per impoundment. The potential marginal benefit that would need to be generated for positive cost benefit values, would consequently be quite low. Rolfe and Prayaga (2007) found that recreational anglers using stocked impoundments in Queensland would be willing to pay an additional $43 per year for a 20% improvement in in their fishing experience. This level of improvement is possible through installation of habitat enhancement structures (Anderson *et al.* 2001, Miranda 2016, Norris 2016, Norris *et al.* 2021). Approximately 50,000-60,000 anglers fish in stocked impoundments each year in Queensland (Gregg and Rolfe 2013, DAF 2021). Therefore, it is highly likely that habitat enhancement in stocked impoundments would deliver positive economic benefits.

Case study 1: Artificial reefs for Florida

Bell (2003) provided a great example of cost benefit analysis covering both new and existing reefs, in a white-paper explaining the results of socio-economic studies of southeast Florida's reef systems. Cost benefit analysis of existing reefs at the regional scale (southeast Florida) estimated annual costs for administering both natural and artificial reefs were to be $32 million, with an annual use value of $751 million (Johns *et al.* 2001). This rendered a BCR of 23.3 with a net annual value-added of $719 million. The study also estimated the upper bound for cost benefit analysis for new reefs in the region, based on the assumption that maximum benefit would be derived from willingness to pay data if congestion was not an issue and use rates remained constant. The exact costs of new artificial reefs was unknown so the author estimated a range of BCR based upon the level to which known state investment was leveraged by matching funds (Table 7). At an investment leverage ratio of 5:1, expenditure across four counties of southeast Florida was estimated to amount to a total of $13.5 million, rendering a BCR of 5.8:1 and value-added of $64.8 million. A more conservative leverage rate of 2:1 for matching funds was estimated to yield a BCR of 11.6 and value-added of $73.0 million. When the cost benefit analysis was performed on existing reefs across southeast Florida, annual costs for administering both natural and artificial reefs were estimated to be $32.3 million, with an estimated annual use value of $750.1 million. This rendered a BCR of 23.3 with a net value-added of $718.6 million.

Reported cost benefits of the Japanese artificial reef and ranching program has received some criticism. In Japan, the average cost of large-scale artificial reef development between 1976 and 1982 was $95 m^{-3} (Mottet 1981). Sato (1985) reported the reefs produced an annual catch of 16 to 20 kg m^{-3} for average-sized reefs for average an annual return of $129–160 m^{-3}, concluding this was evidence of positive economic benefit. However, insufficient data is provided to accurately undertake cost benefit analysis, because the values reported gross production, not marginal benefit, and the costs considered did not appear to include planning, management and other non-construction costs. Ignoring the production that would have occurred in at those sites if the artificial reefs were absent, over-represents the benefits and inflates the cost benefit results. Thierry (1988) suggested that if this information is taken into account, cost benefit analysis would likely produce a negative result.

Economic analysis of FADs in open-access fisheries show that it is unlikely that aggregated fishery profit will improve in the long-term unless restrictions are introduced on fishing effort and catch levels. Overfishing and a decrease in the mean size of fish caught are likely to occur and total yields will decline. These impacts are less likely to occur in underfished or small-scale fisheries where harvest pressure is unlikely to be sufficient to have a significant impact. No evidence of detrimental stock impacts from recreational FAD fisheries were identified. Therefore, the use of FADs for recreational fisheries in Australia has the potential to improve the socio-economic value in a cost efficient manner and deliver positive economic benefits. Further targeted research is required to understand if this is the case and how it may vary between sites across Australia.

Few studies have evaluated the benefits from habitat enhancement of both recreational and commercial fishermen in the same study. Samples (1986) used surveys of both recreational and commercial fishermen to elicit their willingness to pay for FAD deployments in Hawaii. The estimated combined annual benefits ($824,300) were found to exceed the implementation costs ($811,000) only slightly, over a five year period following deployment. This delivered a BCR of 1.02. It should be noted that planning and management costs comprised a significant proportion (38%) of the total project costs, highlighting the importance of incorporating these costs into cost-benefit calculations. However, the benefits calculated by Samples (1986) were likely to be a significant underestimate since two key FAD user groups were not accounted for in the analysis. In the contingent valuation surveys, only recreational vessel owners using the FADs were included in calculations. Recreational vessels had on average 2.6 passengers who would also likely have been willing to contribute to maintain the FAD program, potentially amounting to another 8,000 plus individual contributions. Additionally, commercial pole and line fishers were not included, but surveys around the same time found they realized a 3% increase in annual profits from using the FADs (Sproul 1984). The fleet value of this benefit was estimated at $313,000, raising the total value to $1,137,400. If the line and pole fishery benefits are added to the survey values, the BCR rises to 1.4. If passengers on recreational vessels were also willing to contribute the same amount as vessel owners towards maintaining the FAD program, another $2,050,500 in annual value would have been realized, resulting in a BCR of 3.9. This highlights the importance of identifying and including values from all user groups to reflect the total economic value when undertaking cost benefit analyses.

Costs and benefits need to be comprehensively considered and documented in cost benefit analysis in order to achieve accurate outcomes. As seen above, omitting the benefits to particular user groups can have a significant impact on results. Similarly, the costs of habitat enhancement projects have a significant bearing on the output values of cost benefit analyses, particularly BCR where the cost forms the denominator of the equation. Planning, construction and installation costs can be difficult to calculate, and are also problematic to compare between different regions because raw material and labour costs vary greatly. This is especially the case for structures using recycled/repurposed materials and volunteers for construction and installation. Similarly, ongoing administration and management costs are highly variable and depend upon the habitat enhancement design, durability, structure objective, monitoring and compliance requirements. Nous (2019) estimated that ongoing administration and monitoring costs for NSW offshore artificial reefs amounted to around $104,600 per annum per reef.

In the literature, artificial reefs varied markedly in size, from as small as 0.75 m^2 (Allgeier *et al.* 2013) to a mound of dredged material in the Gulf of Mexico that was 4,800 m wide x 12,400 m long and 6 m high, with a volume of ~ 14 million m^3 (Clarke *et al.* 1988). Ramm *et al.* (2021) calculated that median area of marine artificial reefs was 643 m^2. In Australia, the capital cost of offshore artificial reefs have varied between $1.0 million and $2.3 million for reef ranging in size from 700 m^3 up to 27,000 m^3. At the King Reef, near Exmouth, structures from offshore oil and gas production infrastructure were repurposed to minimise construction costs, whilst still providing necessary reef complexity and volume. This enabled a reef ten to twenty times the size of many concrete or steel offshore reefs to be constructed for a similar cost. The national recommendations for artificial reefs suggest a minimum size of 800-1,000 m^3 for coastal and offshore reefs to be effective (Diplock 2011), therefore new reefs will be looking at capital costs of around $1 million and upwards. The cost of artificial reefs for bays and estuaries in NSW, Victoria and Queensland has generally been lower (*e.g.* $2.65 million for seven artificial reefs in Moreton Bay, DES 2022). This is mostly related to the smaller scale of the reefs (*e.g.* Moreton Bay: 80-300 m^3, DES 2022, NSW: 50-520 m^3, NSW DPI 2022, Victoria: 45-240 m^3, VFA 2022) and lower deployment costs. The lower volume occupied by reefs is typically due to shallow water depths limiting the vertical profile. Unfortunately, no cost benefits analyses were found for artificial reefs in estuarine environments, therefore the impacts of these smaller sized reefs on the return on investment is unclear.

The period over which present cost and benefits are calculated and the rate at which they are discounted also have a significant impact on the outcomes of cost benefit analyses. In Australia, discount rates between 4-7% are commonly used for environmental resource projects (Department of Finance and Deregulation 2007, Walker *et al.* 2008, Hone *et al.* 2022). The period over which the discount is applied is frequently related to the durability of the habitat enhancement structures used and the objectives of a project. However, for habitat enhancement, longer periods are more suitable to ensure full realisation of all fisheries benefits throughout reef aging. For artificial reefs constructed from steel and concrete, 30 years appears to be a commonly stated duration (*e.g.* Nous 2019). For artificial reefs made of natural materials, such as those used in freshwater systems, the discount period may be shorter, or the cost of multiple replenishment events must be included.

Table 7 Summary of cost benefit analyses on fisheries habitat enhancement projects included in the literature review. * indicates capitalised value only because were costs unknown.

Study	Location	Habitat enhancement	Reef scale	Annual use value (million AUD 2021)	NPV (million AUD 2021)	BCR	IRR
Bell *et al.* 1998[1]	NW Florida, USA	Coastal and offshore AR		50.84	1,693.11	131	
Bell 2003	SE Florida, USA	Coastal and offshore AR			4,503.21	23.3	
Bell 2003	SE Florida, USA	New coastal and offshore AR			54.04	5.8-11.6	
Choi 2013	Korea	Concrete AR			35.37	2.66	22.8%
Johns *et al.* 2001	Palm Beach County, USA	Coastal and offshore AR		14.62			
Johns *et al.* 2001	Broward County, USA	Coastal and offshore AR		89.84			
Johns *et al.* 2001	Miami-Dade County, USA	Coastal and offshore AR		16.61			
Johns *et al.* 2001	Monroe County, USA	Coastal and offshore AR		15.04			
Johns 2004	SE Florida, USA	Coastal and offshore AR		5.95	198.26*[2]		
Johns 2004	SE Florida, USA	New AR		1.85	61.60*[2]		
Milon 1988	Dade County, USA	Coastal and offshore AR		55	78.1-572.1		
Nitiratsuwan 1994	Ranong, Thailand	Coastal AR	4,602 m^3		0.07	1.15	15.5%

Study	Location	Habitat enhancement	Reef scale	Annual use value (million AUD 2021)	NPV (million AUD 2021)	BCR	IRR
Samples 1986	Hawaii, USA	FADs			0.013	1.02-1.40, 3.9[#]	
Nous 2019	Port Macquarie, Australia	Steel and concrete offshore AR	1,600 m^3		0.194	1.01	8.8%
Nous 2019	Southern Sydney, Australia	Steel and concrete offshore AR	2,900 m^3		0.223	1.01	9.0%
Pyo 2009	Korea	Coastal AR			5.67	1.29	8.6%
Schug 1978	Florida, USA	Coastal and offshore AR				>1	

[1] The results from Bell *et al.* 1998 were based on absolute, rather than marginal value and therefore the presented decision parameters represent an overestimate. The data has been included in the table for completeness of the review.

[2] The original annual use value and capitalised value reported by Johns 2004 was estimated from a combination of recreational fishing and diving values. Fishing represented 86% of user days, and therefore this proportion of these values is included in the table on the assumption that WTP for angling and diving were similar.

[#] BCR estimated by Sproul 1984 if the value of all persons on vessel accounted for, not just the skipper.

2.4 Summary

This Chapter has shown that the application of habitat enhancement for fisheries management is widespread and has varied regionally. The primary focus is on commercial fisheries in Japan, China, Korea, and Taiwan, small-scale and artisanal fisheries in Europe, Brazil, the south Pacific and south-east Asia, and recreational fisheries in Australia and the USA. The number of locations where artificial reefs and FADs are being installed is increasing. In Australia there is an increasing trend for more structures to be installed, being driven primarily by demand from the recreational sector. There appears to be a high willingness from anglers for licence fee revenue to be invested in the creation of new habitat enhancement sites, using both purpose-built reefs and FADs.

Habitat enhancement can have significant ecological impacts, particularly on the composition and structure of fish assemblages. The installation of artificial reefs typically, increases fish abundance, biomass and diversity when installed at sites where the existing habitat is bare or homogenous. The composition of the resultant fish assemblages is rarely identical to those at nearby natural reefs, but often contains higher abundances of exploitable fish. The higher abundance and biomass of fish species recorded at artificial reefs typically lead to higher catch values for exploited species. These characteristics suggests that purpose-built structures containing traits desirable to exploitable fish have strong potential as a fisheries management tool.

The role of artificial reefs in the both the attraction and production of exploitable fish has historically been debated and been identified as one of the main constraints in greater uptake of artificial reefs by fisheries managers. However, results from recent research clearly demonstrate that purpose-built reefs, constructed of sufficient size and complexity, are capable of both attracting and producing fish of recreational and commercial importance. Succession in the community assemblage influences the relative contributions of each process to the standing fish stocks. Attraction is likely to initially be highest until sufficient epibenthic flora and fauna develop to encourage production to occur. This process can take a decade or more to reach stable, high fish production levels.

The impact of reef installation is most pronounced in species which are habitat-limited at particular stages of their life-history, especially in regions where hard structure is naturally limited. The level of new fish production and existing stock status play a large role in the marginal fishery benefits achieved. Higher productivity levels of exploitable biomass can sustain greater harvest effort without impacting on nearby standing stock.

Despite the high prevalence in the use of artificial reefs and FADs for fishery enhancement, comparatively few studies have quantified the socio-economic benefits derived. Nevertheless, the findings available in the literature suggest that habitat enhancement can generate significant ecological and socio-economic benefits and create substantial economic value. Habitat enhancement is generally considered an economic asset by stakeholders and managers, who are usually willing to contribute towards construction and maintenance. The often high construction cost and attraction to non-resident anglers typically led to high economic outputs for regions where structures had been installed.

While the economic benefits of artificial reefs may potentially be quite large, it is widely accepted that in commercial fisheries the benefits may be offset where an expansion in harvesting pressure leads to overexploitation. Harvest management appears to play a pivotal role in how well fisheries at artificial reefs perform. The comparatively lower fishing mortality from artisanal and recreational fishing efforts, are less likely to have long-term detrimental impacts on fishery stocks at artificial reefs than more intense commercial exploitation. Where a stock is already regionally heavily exploited, or harvest pressure is predicted to be high, locating artificial reefs in protected areas or restricting access to particular harvest techniques may prove more socio-economically beneficial than permitting open-access. The reefs can then act as a source for emigration to other nearby reefs. A more complex version of this approach has effectively been applied in the Mediterranean, where artificial reef complexes consist of two zones: a production zone surrounded by an exploitation zone. Harvest is banned within the more densely constructed production zone, whilst the exploitation zone has a more dispersed format designed to facilitate harvest.

The potential of reef complexes to attract exploitable species was demonstrated to have socio-economic benefits for artisanal and recreational fishermen, but for few examples could be found for larger scale commercial extraction operations outside of overseas ranching projects (habitat enhancement plus stocking). Although cost benefit analysis has generally been lacking, there is a consensus that most artificial reef projects have warranted the expense. However, the economic value of habitat enhancement projects whilst positive, typically returns relatively low benefit cost ratios. Median BCR from the literature covered in this review was only 1.29 (excluding one outlier), and median internal rate of return of 8.55%. Within the limited dataset analysed, the economic value was greater generally for recreational fishery enhancement projects than for those undertaken for commercial fisheries.

The costs for installing artificial reefs in marine areas is typically quite high, with most structures in Australia costing over $1 million to complete. The use of artificial reefs in estuaries, bays and freshwater impoundments has lower costs because the reefs generally consist of multiple smaller modular units, which are cheaper to construct and deploy. Their use in inshore and inland waterways is still being explored, but the initial results have been promising. One potential use of habitat enhancement structures is to attract fish to sites where shore-based anglers have ready access, such as piers, jetties and parks. This improves access to fish and opportunities for anglers, especially those who are mobility limited. This type of enhancement has already occurred in several locations in Australia. This approach has successfully been used extensively in lake and impoundment systems in the USA where it has delivered substantial economic returns for their inland fisheries and become a core management tool. Similar results can be expected in Australia if enhancement is undertaken appropriately, especially in stocked put-grow-take impoundment fisheries where overharvest is less of a concern.

Research and monitoring programs that assess artificial reefs against their goals will be increasingly important in future. This is principally driven by growing environmental awareness and a social licence based on the expectation of rigorous evaluation and environmentally sustainable outcomes. Demonstrating the performance of artificial reefs and FADs against quantitative goals is likely to support this social licence into the future.

Only limited socio-economic studies have been conducted on Australian habitat enhancement projects. There is great opportunity to collect critical socio-economic information on new projects or to evaluate the outcomes of existing ones. One area that urgently needs attention is the use of FADs in recreational fisheries. Despite being extremely popular with anglers who often travel significant distances to use them, little information on their economic value is available. These structures are comparatively cheap to install compared to purpose built reefs, and host a number of highly desired fast growing pelagic species. It is expected that cost benefit analyses of the FAD programs run in most states, will demonstrate substantial positive economic returns. In Queensland, as part of the ‘Switch your fish’ campaign, FADs were installed in an attempt to divert some angling pressure away from heavily-fished demersal species. Understanding the socio-economic and ecological impacts of this project would provide valuable insight into the use of FADs or artificial reefs as management tools. Retrospective evaluation of the King Integrated Artificial Reef in Exmouth presents another great opportunity for confirming the accuracy of the values used in the initial economic appraisal and validate assumptions on important socio-economic parameters. The results would inform and strengthen future economic appraisals for similar reef installation proposals.

Chapter 3. Fish stocking

3.1 General introduction

Stocking of hatchery-reared fish into the wild is now undertaken worldwide in freshwater, estuarine and marine environments to maintain or enhance fish stocks for sport or recreation, to provide food and income, and to conserve threatened species (Cowx 1994, Blankenship and Leber 1995, Welcomme and Bartley 1998b, De Silva and Funge-Smith 2005, Bell *et al.* 2006, Camp *et al.* 2017). Hatcheries breed, rear and release billions of fish annually (Kitada 2018) and the integration of aquaculture and wild fisheries is becoming increasingly recognised as a tool for sustaining and enhancing fishery productivity for capture fisheries (Caddy and Defeo 2003, Taylor *et al.* 2017).

The adequate supply and successful settlement of juveniles into nursery habitat is generally believed to be a major factor limiting fish populations (Doherty and Williams 1988, Munro and Bell 1997, Walters and Korman 1999, Hixon and Webster 2002). Fisheries productivity may therefore be limited by insufficient supply of new recruits or limited juvenile habitat to support new recruits through vulnerable life history stages (Pauly and Christensen 1995). Recruitment limitation may be exacerbated by anthropogenic factors such as fishing or degradation of juvenile habitat (Blankenship and Leber 1995, Blaxter 2000). It is generally accepted that the young stages of aquatic species have much higher mortality rates than older conspecifics due to their small size (Molony *et al.* 2004). Increasing the number of new recruits or assisting them to bypass vulnerable early life-stages when mortality in nature is expected to be high, can improve the level of successful recruitment to a fishery or contribute to natural production in the wild (Waples *et al.* 2016). If adult abundance is recruitment limited, increasing the level of recruitment through hatchery releases might be expected to increase abundance and yield of the recruited stock.

The potential for fish stocking stems mainly from the ongoing development of technology to produce juveniles of a wide variety of species in hatcheries. This technology has paved the way for releasing cultured juveniles into the wild. The availability of large numbers of cultured juveniles has been seen as an opportunity by fisheries managers to potentially reduce the time needed to rebuild some severely over-exploited fisheries or improve the productivity of other healthy fisheries. This can occur through releasing cultured juveniles to restore spawning biomass to levels where the fishery can once again support regular harvests or releasing cultured juveniles to overcome recruitment limitation and increase harvest yields (Munro and Bell 1997, Lorenzen 2005, Bell *et al.* 2006).

Stock enhancement and derivatives of that term have often been used as a generic expression referring to all forms of hatchery-based fisheries enhancement. However, the management approaches to releasing cultured juveniles to help operate fisheries are varied, and clearly defined terminology is necessary to avoid confusion. The current review uses the term *fish stocking* as a generic descriptor for the release or translocation of fish species for fisheries management. Leber (2013) summarized the classifications of stock enhancement employed by Bell *et al.* (2008) and Lorenzen *et al.* (2010) into five main groupings to encourage consistency. We have adopted those definitions:

Ranching - a sustained release program of cultured juveniles into open natural environments for harvest at a larger size in put-grow-take operations. The intent is to maximise production for recreational and commercial fisheries and released animals are not expected to contribute to the spawning biomass.

Stock enhancement - the recurring release of cultured or translocated juveniles into wild populations to augment the natural supply of juveniles and optimize harvests by overcoming recruitment limitation in the face of intensive exploitation and/or habitat degradation yields.

Restocking - time-limited release of cultured juveniles into wild population(s) accompanied by large reductions in fishing pressure, to restore severely depleted spawning biomass to a level where it can once again provide regular, substantial yields.

Supplementation - moderate releases of cultured fish into very small and declining populations, with the conservation aim of reducing extinction risk and conserving genetic diversity.

Reintroduction - short-term releases aimed at restoring populations that have become locally extinct.

Fish stocking has long played a significant role in the management of freshwater systems (Welcomme and Bartley 1998, Paquet *et al.* 2011, Brummett *et al.* 2013) and few new species are being cultured for release. However, increasing global attention is being given to the use of stocking as a management tool in estuarine and marine systems and the number of marine species being cultured for release rapidly grows (Richards and Rago 1999, Halverson 2008, Vega 2011, Camp *et al.* 2017). As in freshwater systems, offsetting increased fishing pressure, habitat degradation and human induced environmental disasters are some of the key drivers of the growing interest in this field (Welcomme and Bartley 1998, Cooke and Cowx 2004, Merino *et al.* 2012, Worm and Branch 2012, Lorenzen *et al.* 2013).

The popularity of stock enhancement stems in part from the perception that this strategy can readily maintain or increase fish population abundance, catches, and fishing effort, and thereby alleviate trade-offs between conservation and socio-economic objectives (Taylor *et al.* 2005, Garlock and Lorenzen 2017). Stock enhancement is popular among some managers and aquaculture producers (Lorenzen 2005) because it is perceived as a straightforward and rapid fix to declining fish stocks (Travis *et al.* 1998). The use of hatchery-reared invertebrates and fish for ranching, stock enhancement or restocking will become more common as the abundances of wild populations continue to decline and market demand for seafoods continues to grow.

Lorenzen *et al.* (2013) argued that the science base for culture-based fisheries enhancements had reached a point where enhancement systems can be effectively designed and their potential contribution to fisheries management goals quantitatively evaluated, thus effectively making such approaches broadly available to fisheries management. Successful stock enhancement depends on cost-effectively releasing well-adapted juveniles in a way that optimises use of the available carrying capacity of the ecosystem to deliver consistent, substantial harvests (Bell 2006). This may give rise to economic or social benefits and provide incentives for active management of fisheries resources (Lorenzen 2008). Stock enhancement and re-stocking following recruitment failure, could lead to faster fishery recoveries, and also be used to supplement natural recruitment to provide a more consistent and higher harvest yield from year to year. Stock enhancement has also been explored in fisheries that are stable and productive but where there is scope for increase in yield. In recreational fisheries, stock enhancement is often seen and promoted as a way of sustaining fish populations, even under very high fishing pressure (Halverson 2008, van Poorten *et al.* 2011). Lorenzen (2014) developed a framework for qualitative design criteria to assist the efficient and effective use of fish stocking for different scenarios (Table 8).

To manage fish stocking sustainably, it is important to understand when to engage in a specific action and when to use alternative management actions (Lorenzen 2014, Arlinghaus *et al.* 2016, 2017, Taylor *et al.* 2017). There has been extensive literature on the issue of the ecological and genetic effects of hatchery releases on wild populations (Cowx 1994, Blankenship and Leber 1995, Hilborn 1998, Hilborn and Eggers 2000, Brown and Day 2002, Aphrahamian *et al.* 2003, Araki and Schmid 2010, Laikre *et al.* 2010, van Poorten *et al.* 2011, Lorenzen *et al.* 2012). However, the social and economic impacts have received much less attention. The threats posed by fish stock enhancement programs, especially introductions of exotic species, are particularly insidious because management recovery tools to overcome some of the adverse effects are not available (Cowx and Gerdeaux 2004).

The focus of stock enhancement research has turned from ‘‘can we release and recapture fish’’ to ‘‘how do we conduct enhancement responsibly?’’ (Ziemann 2004). Any hatchery-based fish stocking initiative should follow the ‘Responsible approach’ outlined in Blankenship and Leber (1995) and further refined by Lorenzen *et al.* (2010). This approach details fundamental stock enhancement principles including evaluation of release densities, examination of ecological processes, assessment of economic performance and development of governance, as well as identifying inherent threats to the system. Although the guidelines were developed for marine stocking, they are constructive and applicable essentially to any species used for stock enhancement in the world.
The updated responsible approach to stock enhancements proposed by Lorenzen *et al.* (2010) contains 15 steps, grouped into three stages.

Stage I: Initial appraisal and goal setting
(1) Understand the role of enhancement within the fishery system
(2) Engage stakeholders and develop a rigorous and accountable decision-making process
(3) Quantitatively assess contributions of enhancement to fisheries management goals
(4) Prioritize and select target species and stocks for enhancement
(5) Assess economic and social benefits and costs of enhancement

Stage II: Research and technology development including pilot studies
(6) Define enhancement system designs suitable for the fishery and management objectives
(7) Develop appropriate aquaculture systems and rearing practices
(8) Use genetic resource management to maximize effectiveness of enhancement and avoid deleterious effects on wild populations
(9) Use disease and health management
(10) Ensure that released hatchery fish can be identified
(11) Use an empirical process for defining optimal release strategies

Stage III: Operational implementation and adaptive management
(12) Devise effective governance arrangements
(13) Define a fisheries management plan with clear goals, measures of success, and decision rules
(14) Assess and manage ecological impacts
(15) Use adaptive management

Although the potential risks of fish stocking are well recognised by researchers and fisheries managers, most stocking still occurs without adequate evaluation of its ecological impacts or cost-

Table 8 Qualitative design criteria for biological-technical components of enhancement fisheries systems serving different objectives. Adapted from Lorenzen (2008 & 2014).

	Ranching	**Stock enhancement**	**Re-stocking**	**Supplementation**	**Re-introduction**
Management aim	Increase fisheries catch	Increase fisheries catch and naturally recruiting stock	Rebuild depleted wild stock to higher abundance	Reduce extinction risk and conserve genetic diversity in small populations	Re-establish populations in historical range
Genetic management	Selection for high return to fishing gear	Selection for high return and separation	Preserve wild population genetic characteristics	Preserve wild population genetic characteristics, maximize effective population size	Assemble diversity of adaptations or use stocks adapted to similar habitats
Release	Early stages / juveniles or large catchable fish, high density	Large juveniles, moderate to high density	Any life stage, high density	Any life stage, low density to supplement natural recruitment	Any life stage, low density
Fishing intensity	High	Moderate to high	Low	Low	Low
Domestication type	Domesticated, mixed	Mixed, wild-like	Wild-like	Wild-like	Wild-like
Developmental manipulations in aquaculture	Sterility, conditioning for natural environment and return/recapture	Conditioning for natural environment and return/recapture, possibly sterility	Conditioning for natural environment	Conditioning for natural environment	Conditioning for natural environment
Wild population	Usually absent	Present (large, but possibly depleted)	Present (depleted)	Present (small, declining)	Absent (locally extinct)

Biological interactions	Interspecific ecological and technical	Intraspecific ecological, genetic and technical	Intraspecific ecological, genetic and technical	Intraspecific ecological and genetic, possibly technical	Interspecific ecological
Auxiliary habitat and environmental modifications	Habitat enhancement	Habitat enhancement or restoration	Habitat restoration, control of non-native species	Habitat restoration, control of non-native species	Habitat restoration, control of non-native species

effectiveness (Pearsons and Hopley 1999, Cowx and Gerdeaux 2004). In addition, limited recruitment or productivity due to poor habitat quality, compensatory changes in growth or survival of fish populations, and a lack of local adaptation of stocked fishes to the wild, can limit the ability of hatchery-reared fish to survive, reproduce and improve the fishery resource (Cowx 1994, Lorenzen *et al.* 2012, Cochran-Biederman *et al.* 2015).

Given the potential risks and benefits, there remains debate among fishery scientists about whether stocking cultured fish and invertebrates is an economically sound way to improve fisheries (*e.g.* Blankenship and Leber 1995, Hilborn, 1998, Blaxter 2000). There are numerous examples in the literature where stock enhancement programs have failed (*e.g.* Moran *et al.* 1991, Amarasinghe 2010), made no discernible impact (*e.g.* Saltveit 2006) or have been highly successful (*e.g.* Drummond 2004, Lorenzen 2008, Hart and Strain 2016). Assessment of the economic and social benefits and costs of fish stocking is required to ensure it is providing an adequate return on investment of limited fisheries management funds. Evaluating the effectiveness of stocking projects is recognised as being fundamental to justifying their expense (Rutledge and Matlock 1986, Blankenship and Leber 1995, Cowx 1998, Lorenzen *et al.* 2010).

Understanding the effectiveness of stocking cultured organisms has been hampered by lack of a scientific, institutional and fisheries-management perspective in planning, design, implementation and evaluation of enhancement programs (Lorenzen *et al.*2010). In early stocking programs, little emphasis was placed on understanding the impact of stocking on fisheries landings, with accountability focused on production and release magnitude (Leber 1999). Despite substantial revenue being spent on fish stocking for more than a century, critical evaluation of stocking practices has only gained momentum more recently (Welcomme and Bartley 1998, Cowx 1994, Leber 2002, Lorenzen 2014). Failure of past stock enhancement attempts on a commercial scale were costly and diverted resources from other management alternatives (Clake and Ianelli 1995, Hilborn 1998).

Fish stocking is a fisheries management tool that particularly requires cost-effectiveness evaluation, due to its common application and heavy investment both economically and socially (Lorenzen *et al.* 2010). Economic investment is seen in the large sums of often public money used to establish hatcheries and annually stock millions of fish in Australia (*e.g.* stocking in NSW costs $40 million per annum, NSW Fisheries 2003). Social investment is seen in the incredible popularity of fish stocking with key stakeholders and the belief that stocking fish is a solution or panacea for fisheries management (Hilborn 1999, Hasler *et al.* 2011, van Poorten *et al.* 2011). Significant trade-offs exist between the cost and benefits of the various biological and socio-economic stocking scenarios (*e.g.* Caddy and Defeo 2003, Lorenzen 2005, Johnston *et al.* 2010, Larkin *et al.* 2011, Leber 2013). Quantitative assessment and modelling of biological and economic dynamics of the enhanced fishery is crucial to the rational evaluation of stock enhancement and alternative or additional management measures.

This Chapter provides an overview on the outcomes from stocking activities for fishery enhancement. The primary focus was to collate and analyse empirical socio-economic feasibility data from Australian studies to enable the outcomes from stocking to be compared against other fishery enhancement options. However, a more global context was utilised due to the limited quantitative socio-economic information available on historic stocking projects in Australia.

First, overviews of national and global fish stock stocking trends are described to provide a context of their extent of use. Second, ecological impacts and risks, including genetic impacts, from releasing hatchery-reared fish are discussed. These have been summarised for Australian projects in numerous workshops and publications already, so only a brief overview is provided (*e.g.* MDBC 2003, Gillanders *et al.* 2006, Burgin *et al.* 2017). Third, empirical studies on the effects of stocking on fishery production and the economic efficiency of stocking programs are summarized and compared between different fishery sectors. Finally, strategic use of stocking in fisheries management is discussed, including how Australian hatchery guidelines can be built upon to improve post-release survival and fitness for hatchery-reared fish, reduce genetic impacts, and improve stocking cost efficiency.

3.2 Trends in stock enhancement

3.2.1 Global trends in stock enhancement

Worldwide, billions of fish from over 300 species are now released annually and the numbers, species and release locations continue to grow (Welcomme and Bartley 1998). Estimates show that more than 160 million juveniles are produced to be ”released to the wild” per day around the world (FAO 1998). Purposeful introductions are particularly common for freshwater systems, with tens of billions of fish introduced yearly into fresh waters worldwide (Halverson 2008, Carrera-García *et al.* 2016, Cucherousset and Olden 2020). Using fish from aquaculture is the most widely practiced tool used for augmenting inland fish production (FAO 1999). In the United States 1.7 billion freshwater fish were stocked in 2004 (Halverson 2008) and more than 40 billion individuals are stocked annually in European freshwaters (Cooke and Cowx 2006).
A significant number of fish are also stocked into marine environments and many countries have now established substantial marine fish stocking programs. According to Born *et al.* (2004), between 1984 and 1997, 64 countries reported some marine stocking activity to the UN Food and Agriculture Organisation (FAO) and approximately 180 species were released. Kitada (2018) identified 187 marine species that were released across 20 countries between 2011 and 2016, with Japan releasing the highest number of species (72), followed by Taiwan (24), the USA (22), China (14), South Korea (14) and Australia (7). Currently over 26 billion juveniles of 180 marine species, including salmonids, are released into the wild every year in more than 20 countries (Kitada 2018). Japan has one of the most advanced stocking programs, annually releasing 76 million juveniles of 37 finfish species and over 3 billion juveniles of 46 invertebrate species into the coastal oceans by partnerships between national and prefectural governments and fishing cooperatives (Imamura 1999, Kitada and Kishino 2006). The Norwegian Sea Ranching Program has been responsible for the stock enhancement efforts of Atlantic cod *Gadus morhua*, salmon and alpine trout (Salmonidae), and European lobsters *Homarus gammarus* since the early 1980s (Svåsand *et al.* 2000). This program has met with only limited success. In the USA, extensive marine stocking programs exist for Red drum *Sciaenops ocellatus* and Pacific salmon species (Salmonidae). In Korea, Taiwan and China, considerable marine stocking is undertaken in conjunction with habitat enhancement to establish and support large-scale marine ranching programs (Bell *et al.* 2008, Kim *et al.* 2017, Lee *et al.* 2018, Yu and Shang 2020).

Fish stocking efforts are conducted for both recreational and commercial fishery purposes, showing regional variation in the primary purpose of activities The primary purposes of stocking in developed countries is for recovery of threatened species and to support recreational fishing, whereas in developing countries, the focus is to increase food fish supplies for rural communities and improve their livelihood through income from fish harvested (Welcomme and Bartley 1998, Ingram and

DeSilva 2015). The successful stock enhancement and ranching programs of some countries including Japan (Kitada 2020), Taiwan (Liao 1999, Su and Liao 1999), Norway (Moksness 2002) and USA (Leber 2002, Rhodes *et al.* 2018), have shown the importance of these activities in replenishment of depleted stocks for commercial and recreational fisheries.

3.2.2 Stock enhancement in Australia

Fish stocking is not a new practice in Australia with fish released regularly for over a century (Gillbank 1996). Early acclimatisation societies stocked local waterways with a range of northern hemisphere fish species, such as salmon and trout (Salmonidae spp), carp *Cyprinus carpio* and other fish to create sport for anglers (Gillbank 1980). Stocking of freshwater native fish began in earnest in the 1970s and 1980s following development of successful breeding and rearing techniques for several native species (Rowland 2013). Hatchery production and stocking of native fish has since increased rapidly, with large numbers of fish now released annually. Hunt *et al.* (2018) suggested this has been driven by the limited number of large species that could be recreationally and commercially targeted (Ebner *et al.* 2016), major modifications to aquatic environments leading to reduced fish abundances (*e.g.* Rowland 1990), and a national love of recreational fishing (Arlinghaus *et al.* 2015).

In a global context, the scale of hatchery-releases in Australia is small (millions of released individuals each year), especially compared with those in China, Japan and USA (billions released). Significant hatchery-based stocking programs have now been established in most states. More than 21 native and 6 introduced freshwater species have been stocked for recreational fishing purposes (Table 9). Additionally, over 15 marine species have been stocked for fishery enhancement purposes, including several commercial fishery trials. In total, over 84 million freshwater fish were reported to be stocked between 2009 and 2015, with recreational species comprising the majority of fish released (Hunt and Jones 2018). The number and variety of marine species being stocked in Australia also continues to grow rapidly. Despite significant public and private investment and the large numbers of fish being released, empirical information on the outcomes of stock enhancement programs in Australia remains very limited.

Table 9 Species currently or historically stocked in Australia for fisheries enhancement purposes

Common name	Scientific name	States
Native freshwater		
Australian bass	*Macquaria novemaculeata*	NSW, QLD, VIC
Barcoo grunter	*Scortum barcoo*	QLD
Barramundi	*Lates calcarifer*	NT, QLD, VIC, WA
Eastern freshwater cod	*Maccullochella ikei*	NSW
Eel-tailed catfish	*Tandanus tandanus*	NSW, QLD
Estuary perch	*Macquaria colonorum*	NSW,
Golden perch	*Macquaria ambigua*	ACT, NSW, QLD, SA, VIC
Jungle perch	*Kuhlia rupestris*	QLD*
Mangrove jack	*Lutjanus argentimaculatus*	NSW, QLD
Mary River cod	*Maccullochella mariensis*	QLD

Murray cod	*Maccullochella peelii*	ACT, NSW, QLD, SA
Northern saratoga	*Scleropages jardinii*	QLD
Redclaw crayfish	*Cherax quadricarinatus*	QLD
Sea mullet	*Mugil cephalus*	QLD
Silver perch	*Bidyanus bidyanus*	ACT, NSW, QLD, SA, VIC
Sleepy cod	*Oxyeleotris lineolata*	QLD
Smooth marron	*Cherax cainii*	WA
Snub-nosed gar	*Arrhamphus sclerolepis*	QLD
Sooty grunter	*Hephaestus fuliginosus*	QLD
Southern saratoga	*Scleropages leichardti*	QLD
Trout cod	*Maccullochella macquariensis*	ACT, NSW,
Native marine		
Abalone	*Haliotis spp.*	NSW*, SA*, VIC*, WA
Black bream	*Acanthopagrus butcheri*	WA
Brown tiger prawn	*Penaeus esculentus*	WA*
Commercial scallops	*Pecten fomatus*	NSW*
Dusky flathead	*Platycephalus fuscus*	NSW, QLD*, VIC
Eastern king prawn	*Melicertus plebejus*	NSW, VIC
Mulloway	*Argyrosomus japonicus*	NSW, VIC, WA
Pink snapper	*Chrysophrys auratus*	SA, WA
Sand whiting	*Sillago ciliata*	QLD*
Saucer scallops	*Amusium balloti*	WA*
Southern rock lobster	*Jasus edwardsii*	VIC*, TAS
Yellowtail kingfish	*Seriola lalandi*	NSW, WA
Western school prawn	*Metapenaeus dalli*	WA
Non-native		
Atlantic salmon	*Salmo salar*	NSW, TAS, VIC
Brook trout	*Salvelinus fontinalis*	NSW, TAS, VIC
Brown trout	*Salmo trutta*	ACT, NSW, SA, TAS, VIC, WA
Cheetah trout	*Oncorhynchus mykiss x Salvelinus fontinalis.*	VIC
Chinook salmon	*Oncorhynchus tshawytscha*	VIC
Rainbow trout	*Oncorhynchus mykiss*	ACT, NSW, SA, TAS, VIC, WA
Tiger trout	*Salmo trutta × Salvelinus fontinalis*	NSW

* Indicates a trial program only

The stocking of hatchery-reared fry and fingerlings has resulted in the creation of valuable new 'put and take' recreational fisheries, as well as the enhancement of existing wild fisheries (Hamlyn and Brooks 1992, Cadwallader and Kerby 1995, Rowland 1995, Holloway and Hamlyn 1998, Rowland 2013, Hunt and Jones 2018). Such stocked fisheries now represent alternative recreational opportunities that simultaneously reduce fishing pressure on marine, coastal and freshwater fish stocks and deliver considerable social and economic benefits. In many instances, if stocking ceased, the associated fishery is likely to decline or collapse, especially in freshwater impoundments where natural recruitment of native fish is rare (Forbes *et al.* 2016).

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 (Taylor *et al.* 2005, Loneragan *et al.* 2013). Stock-enhancement 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 (Hart 2015). More recently, there has been a greater emphasis on the use of bio-economic modelling to inform fish stocking activities and management decisions (*e.g.* Ye *et al.* 2005, Hart and Strain 2016, Taylor 2017).
The outcomes and potential issues associated with fish stocking in Australia have been examined by numerous reviews in recent years. Stocking of freshwater native fishes was reviewed by Harris (2003) and discussed at a workshop on managing fish translocation and stocking in the Murray-Darling Basin (Phillips 2003). Gillanders *et al.* (2006) reviewed the impacts of native fish stocking on fish within the Murray-Darling Basin and recommended that, given the continued increase in stocking of hatchery-reared fish and the potential for such interactions with wild fish, it was essential to take a responsible approach and to monitor and experimentally evaluate any stocking program. The role of stocking in the ongoing conservation and rehabilitation of Murray cod and Golden perch populations was examined by Forbes *et al.* (2016). An Environmental Impact Statement (EIS) on freshwater fish stocking in New South Wales (New South Wales Fisheries 2003) noted numerous considerations were likely to pose a risk to the environment. The EIS also highlighted the lack of specific research into the impacts of stocking on the receiving environment. Hunt and Jones (2018) reviewed the historic extent of stocking freshwater fish species in Australia and assessed whether stocking practices were conducted using a responsible approach (current best practice).

In the Wet Tropics of north Queensland, a review of stocking activities and consideration of the potential impact of fish stocking was undertaken by Burrows (2002). The likely ecological and genetic impacts of Barramundi stocking, and fish stocking in Australia more broadly, were reviewed by Russell *et al.* (2013), who concluded that the releases of barramundi has had little effect on wild populations. Marine stocking in Australia has been reviewed in the NSW Marine Fish Stocking Strategy (2014) and several state-based discussion papers (*e.g.* Borg 2004). These reviews and assessments identified several common threats from stocking including the loss of population genetic diversity, impacts on indigenous aquatic communities (including threats to listed species), and the spread of diseases. Such threats have also been recognised globally and have provided some of the impetus for calls for "responsible fish stocking" (Blankenship and Leber 1995,1997, Lorenzen *et al.* 2010) and these calls have been reinforced in Australia (Taylor *et al.* 2005, 2017). Unfortunately, despite the frequency of reviews into the impacts of fish stocking, very few studies have attempted to assess economic or social costs and benefits for Australian programs.

As the knowledge base on hatchery-rearing and releasing fish in Australia has grown, policy and regulations have also evolved. The focus of policy and management has shifted towards applying a “responsible approach” to fish stocking (refer to page 33) and looking at the ultimate outcomes, rather than concentrating on the number of fish released and the levels of investment required. States and territories have developed a range of strategic policies and regulations reflecting this in order to manage the risks and maximise benefits from fish stocking (Table 10).

Table 10 Key state policies and regulations relating to fish stocking activities in Australia.

State	Stocking policies (incl hatchery guidelines)
Commonwealth	• *National policy guidelines for the translocation of live aquatic animals 2020*
Australian Capital Territory	• *Fish stocking plan for the Australian Capital Territory 2015–2020*
New South Wales	• *NSW freshwater fish stocking fishery management strategy 2005* • *The NSW marine fish stocking fishery management strategy 2014* • *Safe transport of fish and stocking code of practice* • *NSW hatchery quality assurance scheme*
Northern Territory	• *NT fisheries regulations 1992*
Queensland	• *Policy for fish stocking in Queensland December 2020* • *Freshwater stocking and monitoring guideline* • *Broodstock and culture stock collection policy* • *AAQ commercial hatchery code of best practice*
South Australia	• *Policy for the release of aquatic resources 2015*
Tasmania	• *Policy for the translocation of freshwater fish in Tasmania*
Victoria	• *Fish stocking strategy 2021-2025* • *Inland Fish Production and Stocking in Victoria plan.*
Western Australia	• *Policy on restocking and stock enhancement in Western Australia*

3.3 Post-release survival

The survival of stocked fish is one of the most important factors for stocking success (Svåsand *et al.* 2000) and has a large impact on economic feasibility (Hilborn 1988, James *et al.* 2007, Gardner and van Putten 2008a, 2008b, Kitada 2018). The basic premise behind stocking is the assumption that additional recruits will increase stock production by bypassing the recruitment bottleneck that occurs during the high mortality of early life-history stages (Svåsand *et al.* 2000, Caddy and Defeo 2003, Lorenzen 2005, Bell *et al.* 2008). However, if stocking leads to greater densities of fish going through the density-dependent recruitment process, it may not enhance populations as much as expected, and potentially only result in replacement of wild fish (Camp *et al.* 2014). From a biological perspective, to optimize survival and recapture rate, fish should be released when they have reached a size at which they are safe from most predators. However, from an economic perspective, it is the net value of the harvested yield (recreational or commercial) that should be optimized, rather than recapture rates. Therefore, the impacts on wild populations and the cost of hatchery-rearing fish to larger sizes also need to be considered.

Poor survival of hatchery-reared fish is a major concern, as it greatly reduces the efficiency of using hatchery fish to supplement the wild stocks (*e.g.* Hoffman and Bettolli 2005). High mortality after release has been reported in many species (*e.g.* Masuda and Tsukamoto 1998, Salvanes 2001, Kitada 2020) and is thought to be one significant reason why some stock enhancement programs are not resulting in significant economic benefits (Hossain *et al.* 2002). Survival of released fish is not always as high as that of wild fish. For example, survival of hatchery-reared Largemouth bass *Micropterus salmoides* in Taylorsville Lake, Kentucky, was found to be only 74% that of wild fish of the same year class (Buynak and Mitchell 1999). One of the fundamental reasons for this mortality might be that hatchery fish are under selective pressures that increase survival under artificial conditions, but not natural, “wild” conditions (Hjort and Schreck 1982, Cowx 1998). Even small changes in the proportion of released fish surviving to recruitment into the fishery, can result in significant gains in the economic feasibility of stocking projects (Gardner and van Putten 2008b). Unfortunately, measurement of survival has often not received sufficient attention**.**

The anticipated proportion of released individuals recruiting to the fishery and eventually to harvest differs between species (*e.g.* Kitada 2018, 2020). In the literature, post-release survival has been reported for various time frames, making direct comparisons difficult. Survival after the first 24 or 48 hours, first week, first month, season, first year, until maturity or entry to the fishery have all been employed (Table 11). Other studies have estimated instantaneous survival or mortality rates at different ontogenetic stages or do not qualify at what timeframe post-release the survival or recapture value is reported. An important point is that most of this variability results from mortality during the first few months of release, and focus must be on optimising survival at the point of release (Hutchison *et al.* 2012)**.**

High mortality rates are common in released individuals during the first few days post-release (Kristiansen *et al.* 2000, Svåsand *et al.* 2000, Roberts *et al.* 2007, Hutchison *et al.* 2012b). It has been estimated that the mortality rate of released Atlantic cod *Gadus morhua* is at least twice that of wild juveniles (Svåsand *et al.* 1989). Captive-reared Japanese flounder *Paralichthys olivaceus* also show massive levels of mortality in the first few days after release (only 10% make it to 10 days post-release), primarily due to the loss of fish that lack appropriate pigment patterns (Blaxter 2000) and inappropriate antipredator responses (Furuta 1996). These figures are indicative of predator-mediated mortality (Brown and Day 2002). Recapture rates for hatchery-reared fish are therefore typically low, but vary with species and release environment (Table 11).

Despite widespread freshwater stocking having been undertaken for a long time, the majority of studies identified in the literature review dealt with stocking fish and invertebrates into marine environments. Note that small-scale research trials which did not mimic broader stock enhancement projects, were excluded because they often were undertaken in unrealistic environments. A total of 40 studies reporting on post-release fish survival rates for marine stocking were identified, with an overall mean recapture rate of 6.65 ± 5.45%. An additional 23 studies reported on recapture rates for stocked marine invertebrates (mean recapture rate = 11.74 ± 10.19 %), but only 6 studies reporting on 9 species were identified on the recapture rates of stocked freshwater fish (mean recapture rate = 3.30 ± 2.39%). Australian research comprised the majority of the freshwater fish studies and three of the invertebrate studies, but none of the marine fish studies. The majority of studies on marine fish focussed on releases in the large-scale Japanese fishery enhancement program, followed by research from stocking for recreation fisheries in the USA.

Table 11 Posterior mean post-release recapture rates for key groups of stocked species in marine and freshwater environments Key state policies and regulations relating to fish stocking activities

Class	Group	Mean recapture rate
Marine fish	**Overall**	**6.65 ± 5.45%**
	Flatfish	9.76 ± 5.46%
	Red sea bream	8.64 ± 4.76%
	Salmon	1.31 ± 0.81%
	Other	4.73 ± 4.93%
Freshwater fish	**Overall**	**3.30 ± 2.39%**
	River	2.33 ± 2.52%
	Impoundment	4.08 ± 2.22%
Marine invertebrate	**Overall**	**11.74 ± 10.19%**
	Abalone	9.10 ± 6.02%
	Crab	4.69 ± 6.77%
	Lobster	6.00 ± 2.69%
	Prawn	15.73 ± 12.11%
	Scallop	34.50%
	Sea urchin	18.2%
All studies		**7.80 ± 7.52%**

Much of the research into post-release survival of stocked fish has been undertaken in the northern hemisphere, where major long-term stocking programs have been implemented. Northern hemisphere salmonid stocking has been one of the largest stock enhancement programs undertaken, and yet despite almost a century of stocking, typically, less than 5% of all hatchery-reared fish make it to adulthood (McNeil 1991). In the UK, the number is more likely to be below 3% (Salvanes 2001). Similar survival and recapture rates have been reported for released Atlantic cod *Gadus morhua* along the Norwegian and Danish coasts. Mean recapture rates were around 4.0 ± 1.5% (Svåsand *et al.* 2000), but this value may be partially inflated by several substantially more successful examples. The median recapture value of 2% perhaps better reflects the low cumulative recapture rate for hatchery-reared Atlantic cod. These low recapture rates indicate that releases of juvenile cod are unlikely to significantly increase cod production and catches because post-release survival are too low (Svåsand *et al.* 2000).

In Japan, major marine stock enhancement programs have produced slightly more positive recapture results. Kitada (2020) found that on the macro-scale, the Japanese marine stock enhancement program had an overall mean posterior recapture rate of released stock of 8.3 ± 4.7%. Recapture rates have been higher for stocked invertebrates than fish and ranged between 0.9% to 34.5%. Overall, the results suggested that stocking effects were generally small and population dynamics were unaffected by releases, but dependent on the carrying capacity of the nursery habitat. However, comparatively high post-release survival has been reported in some Kuruma prawn (27.6%) and Japanese scallop (35.4%) programs (Kitada and Fujishima 1997, Tanida *et al.* 2003).

Better post-release survival and recapture rates have been reported when juveniles were released in protected waterways, compared to open systems (Svåsand *et al.* 1998a, 2000, Hutchison 2006). In some areas, the difference in recapture rates was almost double (Svåsand *et al.* 2000, Table 12). For example, Australian bass *Macquaria novemaculeata* and Barramundi *Lates calcarifer* recapture rates in Australian rivers are typically <1%, whilst in impoundments recapture rates were 1.4-4.5% (Rimmer and Russell 1998, Hutchison *et al.* 2006, Cameron *et al.* 2012). The higher recapture rates in impoundments could be due to restricted emigration or greater food resources (Hutchison *et al.* 2006, Russell *et al.* 2013).

Although post-release survival estimates from past studies vary significantly between sites and species (Tables 11 and 12), this information can be incorporated in bio-economic models to ascertain whether a stocking project is likely to be economically viable. If the target stock improvement or harvest levels are outlined, the post-release survival rates can indicate the quantity of juveniles that must be stocked to achieve that goal. For example, Loneragan *et al.* (2004, 2006) developed a bio-economic simulation model to predict the feasibility of stock enhancement to produce an additional 100 t of Brown tiger prawn harvest in Exmouth Gulf, Western Australia. Based on post-release survival, it was estimated that a release of 24 million juvenile 1 g prawns would be needed for the fishery to catch about 100 t of released prawns (Loneragan *et al.* 2001a, Ye *et al.* 2005). The production costs for this number of prawns were then estimated to determine economic feasibility of the proposal. The estimated recovery rate used (17%) was based on previous unpublished overseas data, but agrees with the findings in this literature review. Even at this relatively high recapture rate, the model indicated there was a low probability of the project being economically viable. These results also emphasise the potential benefits that could be achieved by improving the post-release survival of hatchery-reared aquatic animals.

An essential objective in the production of hatchery-reared animals is that they should possess similar physical and behavioural capabilities to their wild counterparts in order to minimize differences that would compromise their survival in a natural environment (Cowx 1999, Brown and Laland 2001, Brown and Day 2002). The goal should be for survival of stocked fish to at least match that observed in wild populations of conspecifics (Lorenzen 2000). There are a number of release strategies for maximising the chances of survival of stocked fish including altering size-at-release, release season, release habitat and stocking density (Munro and Bell 1997, Mushiake *et. al.* 1998, Iwamoto 1999, Taylor *et al.* 2009, Hamasaki *et al.* 2011).

Table 12 Recapture rates for hatchery-reared fish

Species	Release period	Location	Fishery sector	Objective	Numbers released	Release size (cm)	Marking method	Recapture rate	Reference
Atlantic cod *Gadus morhua*	1983-1995	Norway	Com	PRS,SAR	1,023,616	8-37	ET, GEN, OM, TC,	2.0% recapture	Svåsand *et al.* 1990, 2000
Atlantic cod *Gadus morhua*	1994-95	Norway	Com	PRS	50,000 offshore 8,000 inshore	19.0-25.2	ET	0.1% offshore 9% inshore after 2 years	Fjallstein & Jâkupsstovu 1999
Barfin flounder *Verasper moseri*	1987 onwards	Hokkaido, Japan	Com	SE	1,000,000 p.a.	8	ET, OM	12.1%	Koya 2005, Murakami 2012, NPJSEC 2015
Black rockfish *Sebastes schlegeli*	1982-1988	Hokkaido, Japan	Com	PRS	150,000	10	FC	0.4-0.75% after 2 yr	Kusakari 1991
Black rockfish *Sebastes schlegeli*	1995-1997	Iwate, Japan	Com	SE	447,394	9.9	FC	11.8%	Nakagawa *et al.* 2004
Black sea bream *Acanthopagrus schlegelii*	1989-1996	Taiwan	Com	SE	56,300	7.1-11.8	ET, FC	0.56%	Liao & Liao 2002
Chum salmon *Oncorhynchus keta*	1974 onwards	Hokkaido & Honshu, Japan	Com	SR	600 billion p.a.	5 g	OM	1.7%	Kitada 2018

Species	Release period	Location	Fishery sector	Objective	Numbers released	Release size (cm)	Marking method	Recapture rate	Reference
Common snook *Centropomus undecimalis*	2000	Florida, USA	Rec	PRS	3,862	7.6-25.1	CWT	6.3% recapture	Brennan *et al.* 2008
Gold-lined sea bream *Rhabdosargus sarba*	1995-1996	Taiwan	Com	SE	18,700	8.7	ET	1.2%	Liao & Liao 2002
Japanese flounder *Paralichthys olivaceus*	1986-1992	Iwate, Japan	Com	SE	611,000	7.9	Latex and brand	14.5%	Okouchi *et al.* 1999
Japanese flounder *Paralichthys olivaceus*	1987	Fukushima, Japan	Com	SE	246,000	10	PTD	15.0%	Kitada *et al.* 1992
Japanese flounder *Paralichthys olivaceus*	1987-1993	Hokkaido, Japan	Com	SE	1,069,000	7–15.5	ET, PTD	10.4%	Ishino 1999
Japanese flounder *Paralichthys olivaceus*	1989	Hokkaido, Japan	Com	SE	149,555	6.0–7.8	PTD, ET, FC	5.7%	Tominaga & Watanabe 1998
Japanese flounder *Paralichthys olivaceus*	1987-1994	Fukushima, Japan	Com	SE	2,762,000	5-10	PTD	17.56%	Fujita 1996
Japanese flounder *Paralichthys olivaceus*	1990-1991	Iwate, Japan	Com	SAR	109,300	4.0-15.0	OM	0.46% recapture	Yamashita 1994
Japanese flounder *Paralichthys olivaceus*	1987-1992	Iwate, Japan	Com	SE	611,000	8.1	Latex and brand	12.7%	Iwamoto *et al.* 1998

Species	Release period	Location	Fishery sector	Objective	Numbers released	Release size (cm)	Marking method	Recapture rate	Reference
Japanese flounder *Paralichthys olivaceus*	1994-2002	Fukushima, Japan	Com	SE	8,260,000	10	PTD	12.1%	Tomiyama *et al.* 2008
Japanese Spanish mackerel *Scomberomorus niphonius*	2002-2003	Seto Inland Sea, Japan	Com	SE	160,122	10.6	OM	15.0%	Obata *et al.* 2008
Japanese Spanish mackerel *Scomberomorus niphonius*	2002-2003	Seto Inland Sea, Japan	Com	SE	145,000 160,200	4 10	OM	0.9-3.1% 7.8-15.8%	Yamazaki *et al.* 2007
Masu salmon *Onchorynchus masou*	1992-1997	Hokkaido, Japan	Com, Rec	PRS	Fry 250,000 Smolt 662,000	0.6-1.0 g	FC	0.22-0.54% fry 0.18-3.5% smolt	Miyakoshi *et al.* 2004
Mulloway *Argyrosomus japonicus*	1996-2004	NSW, Australia	Com, Rec	PRS, SAR	193,800	4.0-8.3	OM	0.08% (0-.21%)	Taylor *et al.* 2009
Pacific threadfin *Polydactylus sexfilis*	1993-1994	Hawaii, USA	Rec	PRS	101,235	4.8-15.0	CWT, VIE	1.7%	Leber *et al.* 1998
Pacific threadfin *Polydactylus sexfilis*	1993-1998	Hawaii, USA	Rec	PRS, SE	430,994	2.5-18	CWT	0.5%	Friedlander & Ziemann 2003
Red drum *Sciaenops ocellatus*	1983-1993	Texas	Rec	SE	140,000,000	2.5-3.5	CWT, FC	<20%	Matlock 1990, McEachron *et al.* 1995,1998

Species	Release period	Location	Fishery sector	Objective	Numbers released	Release size (cm)	Marking method	Recapture rate	Reference
Red drum *Sciaenops ocellatus*	2000-2004	Florida, USA	Rec	PRS	4,027,080	2.5-18	CWT, FC, GEN	0.012%	Tringali *et al.* 2008
Red sea bream *Pagrus major*	1974 onwards	Kagoshima, Japan	Com	SE	0.5-1.3 million p.a.	6.0-7.0	PTD	8.0%	Kitada & Kishino 2006
Red sea bream *Pagrus major*	1977-95	Kagoshima, Japan	Com, Rec	SE	20,000,000-40,000,000 p.a.	6.0	ET2.5-3.5	14% 64-83%	Ungson *et al.* 1993 Kitada 1999
Red sea bream *Pagrus major*	1978 onwards	Sagami, Japan	Com, Rec	SE	0.8-1.2 million p.a.	6.0-7.0	ET, PTD	7.9%	Kitada & Kishino 2006
Red sea bream *Pagrus major*	1987-1988	Oita, Japan	Com	SAR	1,023,000	1.0-4.0	OM	1.5%	Tsukamoto *et al.* 1989
Red sea bream *Pagrus major*	1989	Seto Inland Sea, Japan	Com, Rec	PRS	40,000	10	ET	11.8%	Kitada *et al.* 1994
Red spotted grouper *Epinephelus morio*	2000-2007	Osaka, Japan	Com		4,000 p.a.	10	ET	2.2% (1.5-3.4%)	Tsujimura 2007
Spotted halibut *Verasper variegatus*	1993-2007	Fukushima, Japan	Com	SE	426,704	7.5-51.0	OM, ET	11.1%	Wada *et al.* 2012

Species	Release period	Location	Fishery sector	Objective	Numbers released	Release size (cm)	Marking method	Recapture rate	Reference
Striped bass *Morone saxatilis*	1991-93	Chesapeake Bay, USA	Rec	SE, SAR	31,700,000 400,000	Larvae 4-6	OM	4-23% after 3-5 months 6% to 8 months	Secor & Houde 1998
Striped mullet *Mugil cephalus*	1991	Hawaii, USA	Rec	SAR	90,000	4.5-13	CWT	2.8% recap rate after 11 months	Leber *et al.* 1996
Striped mullet *Mugil cephalus*	1992	Hawaii, USA	Rec	SAR	80,507	4.5-13	CWT	3.1%	Leber *et al.* 1997
Tiger puffer fish *Takifugu rubripes*	1991-2002	Ariake Sea, Japan	Com	SE	1,082,100	0.3–10.2	OM, TC	11.6%	Matsumura 2006
Tiger puffer fish *Takifugu rubripes*	2001-2005	Shizuoka, Japan	Com	SE	452,839	5.6-10.0	VIE	5.1%	Nakajima *et al.* 2008
Turbot *Psetta maxima*	1991-1998	North Zealand, Denmark	Com	SE	437,100	5.3–18.8	ET, OM	3.7%	Støttrup *et al.* 2002
Turbot *Psetta maxima*	1991-1999	North Zealand, Denmark	Com	SE	146,444	5.3–18.8	ET, OM	4.14%	Støttrup and Sparrevohn 2007
FRESHWATER									
Australian bass *Macquaria novemaculeata*	1998-2001	Queensland, Australia	Rec	PRS, SAR	18,000 p.a.	2.0-6.5	VIE	2.7-6.4% recapture	Hutchison *et al.* 2006

Species	Release period	Location	Fishery sector	Objective	Numbers released	Release size (cm)	Marking method	Recapture rate	Reference
								rate after 3 years	
Australian bass *Macquaria novemaculeata*	2007-2009	Victoria/NSW Australia	Rec	PRS, SE	427,000	Fry	OM	0.02% recapture after 1-3 yr	Cameron *et al.* 2012
Barramundi *Lates calcarifer*	1992-1996	Queensland, Australia	Com, Rec	PRS, SAR	6,275-22,902 p.a.	3.0-6.0	CWT	0.26-0.30% recapture	Rimmer & Russell 1998
Barramundi *Lates calcarifer*	1998-2001	Queensland, Australia	Rec	PRS, SAR	18,000 p.a.	2.0-6.5	VIE	0.8-2.0% recapture rate after 3 years	Hutchison *et al.* 2006
Golden perch *Macquaria ambigua*	1998-2001	Queensland, Australia	Rec	PRS, SAR	18,000 p.a.	2.0-6.5	VIE	0.5-3.9% recapture rate after 3 years	Hutchison *et al.* 2006
Mongolian redfin *Culter mongolicus*	2006	China	Com	SAR	230,000 630,000	2.8-9.5	CWT	4.35% 4.65%	Lin *et al.* 2021
Silver perch *Bidyanus bidyanus*	1998-2001	Queensland, Australia	Rec	PRS, SAR	18,000 p.a.	2.0-6.5	VIE	3.8-7.5% recapture rate after 3 years	Hutchison *et al.* 2006

Species	Release period	Location	Fishery sector	Objective	Numbers released	Release size (cm)	Marking method	Recapture rate	Reference
Striped bass *Morone saxatilis*	1981-1996	North Carolina, USA	Com, Rec	SE	53,555	12.8-20.3	ET	6.6%	Patrick *et al.* 2006
INVERTEBRATE									
Abalone *Haliotis* spp.	1980– 1991	Japan	Com	SE	1,261,039	1.7-2.9	GM	12.2%	Hamasaki & Kitada 2008
Black tiger prawn *Penaeus monodon*	1983-1984	Taiwan	Com	PRS, SAR	6,340	30–50 g	ET	2.6%	Su & Liao 1999
Blacklip abalone *Haliotis rubra*	2001-2003	Victoria, Australia	Com	PRS	480	1.5–3.0	n/a	15% after 2 years	James *et al.* 2007
Blue crab *Callinectes sapidus*	2002	Chesapeake Bay, USA	Com, Rec	PRS	25,000	0.6–3.0	CWT, VIE	5-20% survival to maturity	Davis *et al.* 2005
Chinese white shrimp *Penaeus chinensis*	1985-1995	Yellow Sea, China	Com	SE	12.21 billion	4.0–5.0 g	ET, FC	35.6%	Wang *et al.* 2006
Chinese white shrimp *Penaeus chinensis*	1983-1987	China	Com	SE	364.8 million	5.0-7.0	CWT, FC	5.7%	Liu 1990
European lobster *Homarus gammarus*	1983– 2013	UK, France, Norway, Germany, Italy	Com	SE	1,714,947	1 year old	CWT, VIE	4.2%	Ellis *et al.* 2015

Species	**Release period**	**Location**	**Fishery sector**	**Objective**	**Numbers released**	**Release size (cm)**	**Marking method**	**Recapture rate**	**Reference**
European lobster *Homarus gammarus*	1988-1994	Kvitsoy, Norway	Com	SE	128,000	1.2-2.1	CWT	8%	Agnalt *et al.* 2004
Giant cuttlefish *Sepia latimanus*	1992-1999	Nansei, Japan	Com	PRS	372,000	n/a	OM	2.7-7.4%	Oka *et al.* 2004
Greenlip abalone *Haliotis laevigata*	2000-2003	Victoria, Australia	Com	PRS	1,440	1.0-3.0	n/a	35% after 9 months 9% after 3 years	James *et al.* 2007
Japanese scallop *Mizuhopecten yessoensis*	1870s ongoing	Hokkaido, Japan	Com	SR	>3 billion p.a	4.5	None	34.5%	Kitada & Fujishima 1997
Kuruma prawn *Penaeus japonicus*	1970's	Japan	Com	SE	100, million	n/a	CWT, FC	8.4%	Kurata 1981
Kuruma prawn *Penaeus japonicus*	2000	Japan	Com	SE	176,146	0.59	CWT, FC	27.6%	Tanida *et al.* 2003
Kuruma prawn *Penaeus japonicus*	1995-1996	Japan	Com	SE	1,680	3.0	CWT, FC	18.0%	Miyajima & Toyota 2002
Kuruma prawn *Penaeus japonicus*	1980– 1991	Japan	Com	SE	1,261,039	1.7-2.9 g	CWT, FC	12.2%	Hamasaki & Kitada 2006
Mud crab *Scylla paramamosain*	1997– 2001	Kochi, Japan	Com	SE	475,300	0.9-1.5	GEN	0.38%	Obata *et al.* 2006

Species	Release period	Location	Fishery sector	Objective	Numbers released	Release size (cm)	Marking method	Recapture rate	Reference
Pinto abalone *Haliotis kamtschatkana*	2007	Washington, USA	Rec	PRS	11,000	2.6	ET	10.2% after 1 year	Carson *et al.* 2019
Red abalone *Haliotis rufescens*	1995	California, USA	Com, Rec	PRS	50,000	0.8	GEN	<1.0%	Rogers-Bennett & Pearse 1998
Roe's abalone *Haliotis roei*	2014-2016 2011-2013	WA, Australia	Rec	PRS	77,364 9,000 (trans)	>5.0	PTD -	<1.0% 0.24-35%	Strain *et al.* 2019
Short-spined sea urchin *Strongylocentrotus intermedius*	1987– 1998	Hokkaido, Japan	Com	SE	1,961,000	0.8–1.8	PTD	18.2%	Sakai *et al.* 2004
Swimming crab *Portunus trituberculatus*	1998	Shizuoka, Japan	Com	PRS	3,300	2.2	CWT	1.2%	Okamoto 2004

3.3.1 Size-at-release

The size of fish at stocking has been shown to have a significant influence on post-release survival (Svåsand *et al.* 2000, Kelliston and Eggleston 2004), affecting the ability of released individuals to compete for resources, avoid predation or physiologically adapt to a new environment and can ultimately lead to varying levels of survival (Hutchison 1991, Brooking *et al.* 1998, McKeown *et al.* 1999, Hutchison *et al.* 2006). It is generally accepted that the young stages of aquatic species have much higher mortality rates than older conspecifics due to their small size (Molony 2000). The idea of releasing advanced stage pre-recruit or post-recruit (*i.e.* juvenile) fish, instead of larvae or fingerlings, is to protect the fish during their vulnerable early stages when mortality in nature is expected to be high, and instead release them at a larger size when their survival chances have significantly improved (Munro and Bell 1997, Doherty 1999, Ottera *et al.* 1999, Lorenzen 2005, Askey and Johnston 2013). Stocking fish at a size beyond which they are likely to be taken by most predatory fish has often given the best population enhancement results (Hutchison *et al.* 2012). In some instances, only stocked fingerlings larger than certain size thresholds survive until harvestable size (*e.g.* Tsukamoto *et al.* 1989, Masuda and Tsukamoto 1998, Li 1999). However, sometimes fish held for a long time in captivity have very poor predator avoidance and live food foraging skills, and may have poorer survival than conspecifics stocked at smaller sizes. The release of 30 cm trout cod with less than 10% survival (Ebner *et al.* 2006) is a good example of poorer than expected survival in fish stocked at a large size.

There is substantial evidence in the literature for a wide range of species supporting increased post-release survival from stocking at larger sizes (Tables 12 and 13). For example, Hutchison *et al.* (2006) found that fingerling Barramundi *Lates calcarifer*, Australian bass *Macquaria novemaculeata*, Golden perch *Macquaria ambigua* and Silver perch *Bidyanus bidyanus* all had significantly better survival when stocked at 50-65 mm, compared to smaller sizes (20-45 mm). The degree of improvement in survival was closely related to the abundance and composition of predators in the stocked waterbody. Increasing the size of Barramundi *Lates calcarifer* stocked into Lake Awoonga, Queensland, resulted in a nine-fold increase in released fish survival, and underpinned the development of world-class Barramundi impoundment fisheries. Similarly, Hansen *et al.* (1990) found that the survival rate of yearling Rainbow trout *Onchorynchus mykiss* was 24.5-25.5 times greater than for fry, and Hoff and Newman (1995) found that yearling Lake trout *Salvelinus namaycush* had 3-4 times better survival than fingerlings. The recapture rates of hatchery-reared Japanese Spanish mackerel *Scomberomorus niphonius* juvenile stocked in the Seto Inland Sea, Japan, was also found to significantly differ with release size (Yamazaki *et al.* 2007). The smaller 40 mm size release group had recapture rates of only 0.89-3.14%, whilst the larger 100 mm release group had 7.78-15.75% recapture rates. Yeager (1988) demonstrated a size-related benefit in northern Florida where Striped bass *Morone saxatilis* that were released at 150-250 mm were returned 100 times more often than striped bass released at 30-45 mm. In Brown tiger prawns *Penaeus esculentus*, Rothlisberg *et al.* (1999) estimated that releasing 1 million zoea would only result in 100 adults in the wild. By comparison, on growing the prawns in ponds until they were 10 mm length would result in 100,000 adults in the wild if the same number were stocked. Similarly, on growing to 20 mm would result in 200,000 adults in the wild.

However, the size at which fish are released is also governed by economic constraints. For any release program, the value of increased survival at greater length has to be weighed against the added cost of production, because larger fish cost disproportionately more to produce than smaller fish (Russell *et al.* 2004, Hutchison *et al.* 2006, Roberts *et al.* 2007, Hamasaki and Kitada 2008). The increased cost of producing larger fish normally means that fewer fish can be purchased for stocking at the larger sizes. Therefore, to make it worthwhile, any increase in survival rate must be greater relative to the increased cost. A lesser survival rate from a large stocking of small fish may still produce more fish recruited to the fishery than a lower number of stocked large fish with higher survival rates.

A key component of developing successful stocking programs is therefore determining the economically optimal size at which to release fish (Russell and Rimmer 1998, Svåsand *et al.* 2000, Okouchi *et al.* 2004, Hutchison *et al.* 2006). A potential goal is to target the release size to that just slightly greater than that at which size-related natural mortality rates decline rapidly, but which rates of production still allow for substantial numbers of individuals to be produced (Chick *et al.* 2013). The best size at release for economic efficiency (net income/ production and release costs) is sometimes different from the ecologically optimal size at release (Zhao *et al.* 1991, Yamashita and Yamada 1999). Releasing larger fish can result in greater cost-effectiveness if the increase in catch more than offsets the increase in production costs of rearing larger fish (Table 13). However, the balance between increased survival and increased cost of producing stocks for release will determine the overall benefit of stock enhancement at a particular stage.

Table 13 Reported optimal size at release for stocking for various species

Species	**Size range investigated (mm)**	**Optimal size (mm)**	**Reference**
Atlantic cod *Gadus morhua*	80-370	200–300	Svåsand & Kristiansen 1990 Kristiansen 1999
Australian bass *Macquaria novemaculeata*	20-65	50-65	Hutchison *et al.* 2006
Barramundi *Lates calcarifer*	30-300 20-65	300 50-65	Rimmer & Russel 1998 Russell *et al.* 2004 Hutchison *et al.* 2006
Coho salmon *Oncorhynchus kisutch*	4.1-13.9g	13.9g	Miyakoshi *et al.* 2003
European whitefish *Coregonus lavaretus*	88-106	106	Jokikokko *et al.* 2002
Golden perch *Macquaria ambigua*	20-65	50-65	Hutchison *et al.* 2006
Japanese flounder *Paralichthys olivaceus*	40-150	80-110	Yamashita *et al.* 1994 Yamashita & Arataki 2010
Japanese Spanish mackerel *Scomberomerus niphonius*	36-150	100	Yamazaki *et al.* 2007
Largemouth bass *Micropterus salmoides*	55-200	126	Miranda & Hubbard 1994
Lake trout *Salvelinus namaycush*	Fry vs yearling	Yearling	Hoff & Newman 1995

Species	Size range investigated (mm)	Optimal size (mm)	Reference
Masu salmon *Oncorhynchus masou*	15-33g 0.8-32g	33g 32g, but 0.8g most economical	Miyakoshi *et al.* 2001a,b Miyakoshi *et al.* 2004
Mongolian redfin *Culter mongolicus*	28-95	71	Lin *et al.* 2021
Pacific threadfin *Polydactylus sexfilis*	48-150	100-130	Leber *et al.* 1998
Red drum *Sciaenops ocellatus*	60-120 131-375	120 >200	Willis *et al.* 1995 Smith *et al.* 1997
Red sea bream *Pagrus major*	10-40	40	Tsukamoto *et al.* 1989
Silver perch *Bidyanus bidyanus*	20-65	50-65	Hutchison *et al.* 2006
Striped bass *Morone saxatilis*	10-250	150-250	Yeager 1988
Striped mullet *Mugil cephalus*	45-130	85-110	Leber 1996 Leber & Arce 1996 Leber *et al.* 2005
Summer flounder *Paralichthys dentatus*	30-80	75-80	Kellison & Eggleston 2004
Turbot *Psetta maxima*	20-170	<50 and >160	Støttrup and Sparrevohn 2007
Brown tiger prawn *Penaeus esculentus*	Zoea - 20	20	Rothlisberg *et al.* 1999

Several studies have looked at the cost-benefit ratios based on hatchery door prices versus the relative survival rates of the different sized fish stocked. Factoring the known cost per fish produced, by the proportion of the total number produced that are subsequently caught in a fishery, provides a convenient way to examine production cost per yield and determine the most cost-effective stocking size (Leber *et al.* 2005). Pilot studies estimating both production costs and recapture rates for different size classes of released fish are needed to optimize the outcomes of long-term stock enhancement programs. The relative ratios of production cost and survival for each size class can then be compared to identify the most economically efficient release size. Russell *et al.* (2004) conducted this type of experiment to determine the most cost-effective size for stocking Barramundi into open river systems. Although 300 mm juveniles were five times more expensive to produce than 50 mm fingerlings, there was a 29 times greater probability of recapturing the larger fish. Even though the cost of producing larger fish was substantially greater, the improved survival yields a benefit-cost ratio of up to seven times greater than that for smaller fish. In impoundments which are dominated by large Barramundi, recruitment of stocked small size classes of barramundi can be absent or minimal (McDougall *et al.* 2008). It has now become widely accepted that stocking of 300 mm Barramundi into impoundments is the most cost-effective option (Malcolm Pearce, DAF Fisheries manager pers com.). Hutchison *et al.* (2006) investigated the optimal release sizes (20-65 mm) and strategies to maximize survival of released Golden perch *M. ambugua*, Silver perch *B. bidyanus*, Australian bass *M. novemaculeata* and Barramundi *L. calcarifer* fingerlings in impoundments. The results indicated that for all species, stocking at the largest size class examined (50-65 mm) had the highest relative survival rate. This was also the most cost-effective size to stock, unless predator presence in the stocked waterway was low.

Several studies have similarly reported on the benefits of stocking yearlings rather than fry in salmonid fisheries. A study by Hansen *et al.* (1990) found that the survival rate of yearling Rainbow trout was 24.5-25.5 times greater than for fry and more than compensated for the increased production cost. Hoff and Newman (1995) reported that yearling Lake trout had 3-4 times better survival than fingerlings and the costs of each surviving trout in the fishery differed between life stages at which they were stocked. Yearlings actually had an almost 50% lower cost per fish, compared to fingerlings, despite the higher costs of producing and stocking yearlings as opposed to fingerlings. If the objective is to establish or re-establish self-sustaining stocks in river rehabilitation situations, where no population exists, stocking older fish of >2 years old is likely to be more cost effective and lead to faster establishment of self-sustaining populations (Naish *et al.* 2007).

Stocking the largest size juveniles is not always the most economic strategy. For example, Leber *et al.* (2005) found that the most cost-effective size to stock hatchery-reared Striped mullet *Mugil cephalus* in Hawaii was 85-110 mm because the recovery rate was between 2-5 times better than for stocking smaller individuals that were cheaper to produce. Survival rates were even higher for 130-150 mm fish, but the additional production costs were not justifiable. In Japanese abalone *Haliotis* spp., highest survival was observed from releasing 4 cm seedlings, but 3 cm was proposed to be sufficient to avoid most predation by crabs (Inoue 1976, Kojima 1981, Tsukamoto *et al.* 1989). However, economic modelling by Zhao *et al.* (1991) revealed that release at the smaller size of 2 cm was more economical than at 3 cm for some species, due to the additional production costs.

Overall size can make a difference to stocking outcomes. Stocking sizes that hatcheries produce should be based on both feasibility for the hatchery to produce those size classes and research that examines post-release survival of different size classes. Optimum release size for fishery enhancement is ultimately a function of survival, growth rates, hatchery and release costs and the socio-economic value of increased harvest levels and/or stock abundances gained from releasing larger fish (Leber 1995). Pre-release training and behavioural enhancement may be able to help offset the length of time necessary to grow-out fish in order to achieve similar survival rates. This would have the advantage of minimising the development of behavioural deficiencies that accumulate from rearing in captivity.

Larval stocking as a strategy to enhance juvenile production and recruitment has also been suggested as a method for reducing hatchery-rearing deficiencies. Stocked larvae will adopt behaviours that lead to successful feeding, growth, and survival in the natural environments into which they are released, which may be retarded by hatchery-rearing (Secor and Houde 1998). However, the number of larval that would need to be produced and stocked would be extremely large due to the poor survival rates of larvae in the wild (McGurk 1986). No studies were found that examined the effectiveness of this approach.

3.3.2 Behavioural deficiencies from hatchery-rearing

Stocking of hatchery-reared fish does not always result in the increase in fish stocks that might be expected because hatchery-reared fish often have lower survival and provided poorer returns to anglers than wild fish (Brown and Laland 2001, Hutchison *et al.* 2012a). Behavioural deficits linked to domestication effects are among the causes that have been identified. Behavioural deficits may include feeding behaviour (Olla *et al.* 1994, Ellis *et al.* 2002) predator avoidance behaviours (Malavasi *et al.* 2004, Ebner *et al.* 2006), movement or dispersal (Bettinger and Bettoli 2002, Ebner and Thiem 2006) and territorial behaviours (Metcalfe *et al.* 2003). The first two behavioural deficits, feeding and predator avoidance are those where the most effort has gone into minimising domestication effects to improve survival and therefore the value of hatchery-reared fishes. Examples of feeding and predator avoidance deficiencies are given below for fish, crustaceans and molluscs, along with strategies that have been applied to overcome these deficits.

3.3.2.1 Feeding behavioural deficits and solutions

Inefficient foraging is considered a major deficit for hatchery-reared fish that are released into the wild (Donadelli *et al.* 2015). For example, hatchery-reared Turbot *Scophthalmus maximus* fed on a diet of pellets were less motivated than wild conspecifics to feed on Shrimp *Crangon crangon.* Hatchery-reared Turbot were also less efficient at capturing Shrimp and took smaller prey than wild conspecifics (Ellis *et al.* 2002). Hutchison *et al.* (2012b) found that sub-adult long-term pellet reared Murray cod *Maccullochella peelii* would not take live shrimp feeds. A comparison of hatchery-reared and wild Steelhead trout (sea-run Rainbow trout) *Onchorynchus mykiss* found that hatchery-reared smolts were less piscivorous than wild smolts and hatchery-reared non-migratory (residual) fish exhibited more surface feeding behaviour than wild parr (Simpson *et al.* 2009). A further example comes from a comparison of hatchery-reared and wild Japanese flounder *Paralichthys olivaceus* (Furuta 1998). Wild flounder showed rapid feeding behaviour, quickly returning to the bottom after taking a prey item, whereas hatchery-reared flounder spent longer in the water column, where they would be more vulnerable to predation. Hatchery-reared flounder also showed more frequency in settling behaviour, rather than staying close to a single spot.

Duration in captivity may influence the ability of stocked fish to switch effectively to wild diets. For example, Grausgruber and Weber (2021) found that Walleye *Sander vitreus* stocked as fingerlings had higher average proportions of empty stomachs and consumed more benthic invertebrates, but less fish, than walleye stocked as fry. The stocked fingerlings consumed lower quality prey for at least 49 days post-stocking. Norris (2004) found duration of pellet rearing influenced the speed at which Trumpeter whiting *Sillago maculata* could locate pellet prey. Fish reared on live prey relied more on visual cues, while those reared on pellet diets relied more on olfactory cues. Long-term pellet rearing also resulted in physiological changes that caused an increase in the number of external taste receptors, except in the gular region.

For some species at least, hatchery rearing can lead to poor foraging or feeding behaviour in fish stocked into the wild. However, there are strategies that can be used to help overcome these foraging deficits. For example, rearing fish larvae and fingerlings in semi-natural conditions, such as ponds (rather than tanks) where they are exposed to live prey items like zooplankton and chironomid larvae (rather than pellet feeds) can lead to more natural foraging behaviours. Olson *et al.* (2000) compared the survival of intensively reared large (120-140 mm) Walleye *S. vitreus* fingerlings and extensively reared small (30-50mm) fingerlings stocked into four small lakes. Intensively reared fish were raised in hatchery tanks, beginning on a diet of brine shrimp, then switching to pellets. Extensively reared fish were raised in 0.3 to 0.5 ha earthen ponds and fed on zooplankton species. In two lakes, survival of pond reared fingerlings was better than the larger intensively reared fish. Pond reared fingerlings were larger than the intensively reared fingerlings by the end of the growing season, suggesting better foraging efficiency.

McKeown *et al.* (1999) examined the interaction between stocking size and rearing method on post-stocking survival of Muskellunge *Esox masquinongy* in New York State. Greater length at stocking led to better survival, but after accounting for length at stocking, pond reared fish had better survival than trough reared, pond finished fish, which in turn had better survival than totally trough reared fish.

Fuss and Byrne (2002) compared survival from smolt stage to adult, for Coho salmon *O. kisutch* extensively reared in a semi-natural rearing pond, containing large woody debris and rock to fish reared in conventional ponds. Density of fish in the semi-natural pond was only 5% that of fish reared in conventional hatchery ponds. Conventionally-reared fish were fed entirely on pellet feeds, whereas fish in the semi-natural pond had access to wild feeds and pellets. Survival of smolts to adulthood was higher in fish reared in semi-natural ponds, than by conventional methods. However, the increased survival did not offset the increased adult yield that would have been realised from standard hatchery production techniques. Nevertheless, Fuss and Byrne (2002) conceded that semi-natural rearing methods would have value for fish to be released in recovery programs.

Many native fish fingerlings reared for freshwater fish stocking in south-eastern Australia are reared in ponds where they feed on live food (Hutchison *et al.* 2012a). If stocked soon after pond harvest, it is likely that these pond reared fingerlings may be quite well equipped for recognising and foraging on live prey items in the wild. In contrast Barramundi *L. calcarifer* are reared on artificial feeds from an early stage of life, after a short period in green-water culture.

For those fish that are reared on manufactured diets there are options to retrain fish to take live feeds, especially younger fish. A review by Hughes *et al.* (1992) provided evidence that fishes can optimise foraging behaviour through learning. Warburton (2003) presented further evidence for learning of foraging skills by fish. Brown and Laland (2001) reviewed research into social learning in fishes. They presented unequivocal evidence for social learning (learning from conspecifics) in fishes. Foraging skills can be retained for a short time after learning. For example, foraging skills are retained for up to three weeks post-learning in Fifteen-spine sticklebacks *Spinachia spinachia* (Croy and Hughes 1991a, b). Over recent decades, research has been undertaken, including life-skills training programs, to develop more wild-like behaviours in hatchery produced animals (Näslund 2021). Virtually all life-skills training relating to foraging in aquatic species has been conducted on fish and not invertebrates. This could be an effect of foraging not generally being perceived as a problem in invertebrates, many of which are generalists, grazers, or use modes of feeding such as filtration (Näslund 2021).

With one or two exceptions (*e.g.* Masee *et al.* 2007), Näslund (2021) noted most studies show positive effects of live food exposure on later foraging success. However, many of these studies have been done in a laboratory setting without any post-stocking evaluation (*e.g.* Ellis *et al.* 2002, Jackson *et al.* 2013, Donadelli *et al.* 2015). Studies show that for some species, exposure to live feeds can improve foraging with increased experience, but better results have been achieved when combining foraging training with an enriched environment (Brown *et al.* 2003). Enrichment may include installation of timber, rocks, plastic tubing, PVC structures, live and plastic plants for habitat complexity (Brown *et al.* 2003, Zhang *et al.* 2019). Enriched environments have been shown to improve cognitive abilities in fish which may benefit their survival (Strand *et al.* 2010).

There are not many examples of where the effects of foraging and live feed training in the hatchery have been quantitatively examined after stocking into the wild. However, several studies have demonstrated some success. Szendrey and Wahl (1995) examined post-stocking survival of minnow fed and naïve pellet fed Muskellunge *Esox masquinongy* and Tiger muskellunge (Muskellunge x Northern pike *E. lucius*) stocked into reservoirs. The minnow-reared fish had better post-release survival than the pellet-reared fish. Szendrey and Wahl (1995) attributed this better survival to colour differences between the two groups rather than to foraging efficiency. They believed the colouration of the pellet fed group made them more vulnerable to predation.

Czerniawski *et al.* (2015) trained European grayling *Thymallus thymallus* fry to feed on daphnia and chironomid larvae for 7 weeks. A control group was reared on pellets. These fish were then released into enclosed sections (2 km) of two streams. Post-stocking survival of the live-feed group after three months at large ranged from 22% to 28%, whereas post-release survival of the pellet-fed group ranged from 10% to 15%. Thus, survival in the live-feed trained group was double that of the pellet-fed group. This represents a substantial increase in potential value to the trained group. The live feed trained group also achieved a higher body mass post-stocking and had higher levels of invertebrates in their stomachs than the pellet fed group.

Czerniawski *et al.* (2011) compared the post-release survival of Atlantic salmon *S. salar* fry and sea-run Brown trout (Sea trout) *S. trutta* fry reared on either zooplankton, live nekton fish larvae or on pellet feeds. The hatchery feeding phase lasted for ten weeks. The fish were then stocked and at large from May 2009 to September 2010 in a stream divided into three replicate sections. Mean survival of the pellet-fed group of Atlantic salmon was 10%, mean survival for the nekton-fed group was 46% and for the zooplankton-fed group 53%. For the Sea trout, mean survival of the pellet-fed group was 11.3% with survival in the nekton-fed and zooplankton-fed groups 25% and 61%, respectively. In both species (Atlantic salmon and Sea trout) survival of the zooplankton-fed group was over 5 times that of the pellet-fed group, representing a major value-add to the released fry. Specific growth rate of the zooplankton-fed groups was also greater than growth in the pellet-reared group in both species. The specific growth rate for the zooplankton fed Atlantic salmon and Sea trout was 2.92 and 3.38 respectively, whereas for the pellet fed group of both species specific growth rate was 2.05 and 1.99.

In contrast to the above study, Costas *et al.* (2013) did not find significant differences between live prey-trained and control Atlantic salmon condition post-release in the wild, with both groups having relatively poor condition. This study used a much shorter live food training period (two weeks) than Czerniawski *et al.* (2011, 2015).

More research needs to go beyond the laboratory and into the field to examine the impact of live food foraging training in the hatchery. The results from the limited number of studies that have done this, suggest that such training has the potential to significantly add value to stocked fingerlings through increased rates of survival and better growth rates. The extent of the benefit is likely to vary among species, stage of life trained and duration of training.

Pond rearing on live feeds and habitat enrichment also show promise as means to value add to hatchery-produced fish to be used for stocking. This can be achieved through increasing post-release survival and growth rates relative to conventionally tank reared fingerlings, although more field-based validation is required. There is very little information available on the economics of foraging training and whether this adds significantly to the cost of hatchery production. Research into the cost and benefits of foraging training is therefore required. Additional production costs will need to be offset by sufficient increases in survival for this to be an economically feasible approach. Current pond production methods for native freshwater fish fingerlings produced in Australia probably already meet many of the requirements for producing fingerlings that are effective foragers, so in their case, there may be little change in production costs if any further modifications are made, such as habitat enrichment in the ponds.

3.3.2.2 Predator avoidance behavioural deficits and solutions

One of the greatest causes of loss of stocked hatchery-reared fish is predation (Olla *et al.* 1994). Most mortalities occur within the first few days after stocking (Olla *et al.* 1994, Brown and Laland 2001, Sparrevohn and Stoetrupp 2007). In a stocking experiment in a Texas lake, Buckmeier *et al.* (2005) estimated that 27.5% of largemouth bass fingerlings (*Micropterus salmoides*) were taken in the first 12 hours post-stocking. Hutchison *et al.* (2006) found that variation in predation rates on different batches of fingerlings released on the same day were reflected in the recapture rates of the same batches 12 months later.

Hatchery-reared animals are usually reared in predator free environments. There may be some exposure to cannibalism in some species, or to limited bird predation in un-netted ponds (Hutchison *et al.* 2012a), but in general most hatchery-reared animals are naïve to predators. According to Brown (2003), many prey species of fish do not show innate recognition of potential predators, but acquire this skill based on the association of alarm cues with the visual or chemical cues of the predator. While some reactions may be innate (Kelley and Maguran 2003), hatchery-reared animals often show a different reaction to their wild conspecifics in the presence of a predator, which suggests that anti-predator responses are at least partly learnt.

Various studies have shown that hatchery-reared animals are more vulnerable to predation than their wild counterparts. For example, hatchery-reared sea ranched salmonids are less afraid of predators and have lower survival (Petersson and Jarvi 1999). Ebner and Thiem (2006) found that 95% of wild radio-tagged Trout cod *Maccullochella macquariensis* were alive 13 months post release, but only 9% of tracked hatchery-reared conspecifics survived over the same period. Evidence suggests most of this mortality was from cormorant predation (Ebner *et al.* 2006). Yamamoto and Reinhardt (2003) found that hatchery-reared Masu salmon *O. masou* were more willing to leave cover and feed under chemically simulated predation risk than wild Masu salmon, indicating reduced predator avoidance in the hatchery-reared fish. Similarly, hatchery-reared Grass carp *Ctenophayngodon idellus* have been shown to be predated more readily than their wild counterparts (Tang *et al.* 2017).

As noted for foraging behaviours above, there is evidence that fish can be trained. Various studies have used a range of techniques to try to train fish and aquatic invertebrates to recognise and avoid predators (Brown and Laland 2001, Näslund 2021, and references therein). Kelley and Magurran (2003) provided evidence that visual predator recognition skills are largely built on unlearned predispositions, whereas olfactory recognition typically involves experience with conspecific alarm cues. They also found that populations vary in their capacity to learn.

Some predator recognition training studies have met with success and others have not. Differences related to duration of training, the training method and the species used could explain the varied results. Examples of various studies and their outcomes are given below. Most examples given are for fish, but there are some studies that have also investigated reducing predation risk for hatchery-reared crustaceans and molluscs.

Some attempts at training fish to avoid predators were not very successful. In an early attempt, Fraser (1974) used an electrified plastic model Loon (a piscivorous bird) moving through a hatchery raceway to train Brook trout *Salvelinus fontinalis*. Fish close to the Loon received an electric shock, with an effective electric field being produced 10-12 cm from the head of the model. Following training, untrained control fish and trained fish were released into a small lake. However, mean survival of trained and untrained fish did not differ, suggesting the training technique was not effective. The failure was attributed to the fact that during training fish only had to move 50 cm to the side to avoid being shocked, whereas a real predator would turn and chase, but the fish had no experience of this.

3.3.2.2.1 Laboratory or tank-based studies

Various predator avoidance training programs have met with some success under laboratory conditions with changes in predator avoidance behaviour confirmed in tank or pond trials (*e.g.* Järvi and Uglem 1993, Brown *et al.* 1997, Arai *et al.* 2007, Walsh *et al.* 2013*).* Some studies have also demonstrated improved survival of predator trained animals in the wild. As for foraging training, lab-based or tank-based experiments are numerous, but far fewer experiments have evaluated predator-conditioned hatchery-reared fish and invertebrates following release into the wild.

Petersson *et al.* (2014) had a partially failed conditioning experiment with 0+ Brown trout *S. trutta*. Predator-naïve hatchery-reared Brown trout and wild Brown trout were assessed in behavioural trials that lasted for eight days. Predator-conditioned hatchery-reared brown trout were also assessed. Predator conditioning consisted of holding 0+ fish in a stream-water aquarium with adult Atlantic salmon and adult Brown trout for two days. Predator conditioning did lead to increased use of shelter by conditioned fish compared to naïve fish, but did not lead to increased use of time in a predator free area compared to naïve fish. It is possible that the training period used by Petersson *et al.* (2014) was too short. Hutchison *et al.* (2012a) found at least three days of training was required to elicit a significant change in predator response under laboratory conditions from hatchery-reared trained Silver perch *B. bidyanus* and Murray cod *M. peelii* fingerlings compared to naïve fish, but only 48 hours of training was required to get a significant response from Freshwater catfish *Tandanus tandanus* fingerlings. More research needs to be directed towards optimal predator avoidance training periods for different species and life stages of fish. Such research should investigate the cost efficiency and effectiveness of the various approaches to optimise economic performance.
Common whelks *Buccinam undatum* exposed to predatory starfish odour and alarm cues, with intermittent exposure to live predators over two months, did not show notably improved antipredation responsiveness (Justome *et al.* 1998). However, a related study that also used contact with predators and potential alarm signals from conspecifics, did elicit an improved predator response from common whelks (Rochette *et al.* 1998).

In a lab-based experiment, exposure of hatchery Rainbow trout *O. mykiss* to alarm cues from conspecifics in combination with kairomones (chemical odours) from Brown trout *S. trutta* predators led to a reduction in time spent exploring and an increase in time spent frozen, suggesting appropriate responses to predation were being learned (Kopack *et al.* 2015). However, this experiment did not proceed to post-release field trials in the wild.

Walsh *et al.* (2013) exposed hatchery-reared juvenile Japanese flounder *P. olivaceus* to predators by placing them in predator free cages in the wild for 6 days. The effects of this treatment were then examined by tank-based predator response experiments. Cage-conditioned fish exhibited a predator response approaching that of wild flounder, which was a better response than that of hatchery-reared naïve fish.

Vilhunen (2006) exposed hatchery-reared Arctic charr *Salvelinus alpinus* fingerlings to the odour of Pike perch (Zander*) Sander lucioperca* fed on Arctic charr. Fish were conditioned once or four times, with a four-day gap between each conditioning event. In tank-based tests, trained fish showed better spatial avoidance in the presence of pike perch than did naïve hatchery-reared fingerlings. Conditioning just once was sufficient to elicit a significant change. Vilhunen (2006) suggested there could be economical and ethical advantages of training with chemical cues combined with high reliability.

In a study on hatchery-reared Scallops *Argopecten purpuratus* both juvenile and adult scallops were exposed to continuously to Starfish *Meyenaster gelatinosus* odour with direct predator contact for 30 minutes three times per day. Training lasted for three days. Post-training reaction times of juveniles to predators was 25% faster, and the improvement was 50% faster for adults compared to untrained controls. Escape behaviour (clapping rate and clap duration) also increased in trained juveniles (Brokordt *et al.* 2011).

In an experiment with hatchery-reared juvenile European lobsters *H. gammarus* an alternative approach was used to reduce susceptibility to predation. Rather than expose the lobster juveniles to real predator cues, the juvenile lobsters were conditioned to use small round clay pot shelters (van der Meeren 2001). Some mock predatory cues such as touching were used to cue some lobsters to use shelters during the training period. In subsequent testing in tanks, trained lobsters were quicker to use shelters than untrained lobsters and time to reach shelters was less in touched lobsters. These results suggest trained lobsters may be better able to avoid predators, although this was not field tested.

Agnalt *et al.* (2017) used a similar approach to van der Meeren (2001) to improve the survival of juvenile European lobsters. They used two treatments to raise newly hatched post larvae. One batch were reared in single compartments, whereas another group were reared communally in tanks with sand substrate and shelters to allow development of burrowing and sheltering behaviours. The conditioning period was at least 8 months. In a second experiment four-month-old lobsters were purchased from a hatchery and divided into the same two treatment groups, but conditioning in the second experiment only lasted 47 days. At the end of the conditioning experiments juvenile lobsters of both groups were released into semi-natural conditions with provision of scallop shell shelters. Survival rates of trained lobsters were three to four times higher than untrained lobsters after 45 days.

In a study using direct contact training for Coho salmon *O. kiisutch*, predator trained fish had similar survival to naïve fish (75% and 71% respectively) when tested together in a laboratory predation trial, but when tested alone, the naïve fish only had 46% survival (Patten 1977). This suggests a social learning component may assist survival. It is therefore possible that training only a proportion of fish that are to be stocked may be sufficient to assist the whole batch post-stocking, especially if the fish are at a life history stage that gathers in groups (shoals).

3.3.2.2.2 Field testing of trained animals

Hutchison *et al.* (2012a) exposed hatchery-reared Silver perch *B. bidyanus* and Murray cod *M. peelii* fingerlings en-masse in large tanks to predators (Murray cod sub-adults, adult Golden perch *M. ambigua* and large adult Spangled perch *L. unicolor*) behind a screen for three days. Skin extract from either Silver perch or Murray cod (the prey species) was released twice daily on the predator side of the screen as an alarm cue. Both species showed improved post-training responses to predators in lab-based tank trials compared to naïve untrained fish. After release into the wild, mean survival of trained Murray cod was double that of untrained naïve fingerlings, and in locations where predator densities were high, survival was up to four times higher for the trained Murray cod than for the predator naïve fish. Trained Silver perch and untrained Silver perch had no significant difference in survival in the wild post-stocking. Based on tagged recaptures, it was found that trained and untrained silver perch had formed mixed schools within 24 hours of stocking despite being stocked 1km apart. It is possible that social learning from trained silver perch may have assisted the survival of the untrained Silver perch.

D'Anna *et al.* (2012) conditioned hatchery-reared White sea bream *Diplodus sargus* in tanks to a predator (Conger eel *Conger spp.*), to a shelter or both, with a fourth group of fish having no conditioning as predator-naïve controls. A total of 1,500 fish were conditioned in each group. Conditioning lasted 30 days. All conditioned fish were marked with tags and after the conditioning period were released into the sea. Post-release observations of tagged fish showed that survival of predator-conditioned fish was almost twice that of the predator-naïve fish. This result is similar to that of Hutchison *et al.* (2012a). Shelter-conditioned fish dispersed shorter distances than naïve fish, reducing their exposure to predation.

Experiments by Lönnstedt *et al.* (2012) exposed predator-naïve juvenile Damsel fish *Pomacentrus wardi* to olfactory and/or visual cues of common benthic predators. Fish were also exposed to high or low feed rations. After training, fish were released on a reef in the field and allowed to settle. Well-fed fish conditioned to visual, chemical or a combination of predator cues survived eight times better than untrained fish in the first 48 hours after release. This represents a considerable gain in survival from simple training techniques combined with good nutrition.

Florida bass *Micropterus floridanus* conditioned to predators in outdoor ponds had almost double the post-stocking survival of Florida bass reared in indoor raceways and fed on pellets (Trippel *et al.* 2018). However, even pond-reared bass not exposed to predators had better post-stocking survival than those reared in raceways. Improved survival seems to be related to a combination of outdoor pond rearing and predator exposure.

Not all experiments with predator-conditioning have been shown to deliver a significant improvement on post-release survival. Archer and Crowl (2014) suggest the lack of demonstrated benefit for trained fish stocked into the wild may be because they do not retain memory of novel predators for extended periods without ongoing reinforcement. For example, June sucker *Chasmistes liorus liorus* retained learning for at least two days, but lost it by 10 days after initial exposure. In the case of the successful Murray cod *M. peelii* stocking experiment (Hutchison *et al.* 2012a), fish were transported and stocked immediately after the three-day training period, and this may have negated any potential memory loss effects.

Berejikian *et al.* (2000) suggested a problem with past efforts to assess the effects of training fingerlings on the success of field releases, is that both trained and untrained fish have often been released together. This enables the control fish to rapidly acquire anti-predator behaviour from the trained fish through social learning (see also Hutchison *et al.* 2012a Silver perch example), but the improved survivorship of the control fish offsets the apparent effect of the training by reducing differences in mortality between test and control fish.

3.3.2.3 Is it worth investing in hatchery-based predator conditioning

There is no doubt that for some species of hatchery-reared animals, predator conditioning training can lead to substantial post-release improvements in survival. Studies cited above had improvements in post-release survival ranging from two to eight times that of untrained controls. None of the above studies have included an economic assessment of the training techniques in terms of added cost to the hatchery versus improvement in survival post-release. If the predator training techniques do not result in substantial additional time and labour (and therefore cost) or a substantial reduction in the output of hatchery-reared animals, then any additional costs would be more than compensated for by the increased survival and increase the economic feasibility of stocking projects.

For example, Hutchison *et al.* (2012a) demonstrated that Murray-cod *M. peelii* could be trained en-masse over three days for a doubling in post-release survival. An additional three days in the hatchery for training is unlikely to double the cost of the fingerlings, so any minor cost increase in the fingerlings (*e.g.* 10-20%) would be more than offset by the improved returns post-stocking. D'Anna *et al.* (2012) also trained fingerlings en-masse, but over 30 days. If this training was in conjunction with the normal rearing period, then there may not be much addition in cost to the fingerling production. However, if the 30-day training period was in addition to the normal rearing period, then the benefits of the additional cost for a doubling in survival may be questionable. Research to identify the minimum period required for an effective improvement in post-stocking survival may be required for different species to maximise the cost-benefits of introducing predator conditioning to hatchery procedures.

3.4 Impacts of stock enhancement

The potential for stocking is primarily related to the population dynamics of the species within a given ecosystem, economic cost-benefits, fisheries management and socio-economic impacts. There is a huge body of literature detailing the cost, benefits and risks associated with stocking cultured fish. Fish stocking is often contentious due to its high investment, limited scientific evaluation, and typically divided opinion from key stakeholders (Hunt and Jones 2018).

Much of the literature has focussed on the ecological effects on target and non-target species resulting from the introduction of hatchery-reared or translocated fish. Although stock enhancement seems straightforward and its success probable, there are many issues that can cause hatchery stocking efforts to trigger permanent harm to the target population rather than deliver the expected conservation benefits. There is evidence suggesting negative effects of hatchery rearing on a variety of fish species in past supplementation stocking programs (*e.g.* Cowx 1994, Welcomme and Bartley 1998, Hilborn 1999, Brown and Laland 2001, New South Wales Fisheries 2003, Lorenzen 2014, Kitada 2018, Cucherousset and Olden 2020).

Conversely, there has been far more limited quantitative evidence describing the economic feasibility of stock enhancement. Substantial revenue has been spent on fish stocking for over 100 years, but critical evaluation of stocking practices has only gained momentum more recently in the last two decades (Leber, 2002, Lorenzen 2014). The potential for stocking is not derived solely from whether or not the species can be cultured in sufficient quantities relative to the magnitude of the natural recruitment, but instead from the resultant benefits to the associated fisheries. A major problem in justifying the expense and effort associated with stock enhancement, is determining if it is successful. Leber (1999) points out that success has typically been measured by production levels and numbers of fish stocked. The emphasis on production as the principal measure of success has been maintained because after hatchery-reared fish are released, it has been difficult to track the stocked fish or to distinguish them from wild fish.

Advances in the technology available to distinguish released fish from wild fish have enabled the fate of these fish and their contribution to fisheries to be better understood. Significant benefits from stocking can occur under certain conditions, resulting in increases in fisheries yield, the rebuilding of populations, and the partial mitigation for habitat loss and ecosystem effects of fishing (Lorenzen 2008, Cowx 1999). A clear understanding of the socio-economic impacts of stocking will enable any benefits to be compared to the potential negative ecological effects and highlights why evaluating the effectiveness of stock enhancement projects is recognised as being fundamental to justifying their expense (Cowx 1999).

In the following sections, a brief overview is provided on the ecological effects of stocking fish, followed by a summary on socio-economic impacts resulting from fish release. Together, these sections will help inform the discussion on whether stock enhancement is socio-economically beneficial, and if the benefits can outweigh any negative ecological impacts.

3.4.1 Ecological impacts

A wide range of ecological and environmental risks have been recognised to be associated with fish stocking activities (Cowx 1998). The effects of stocked fish on the pre-existing wild population, other aquatic species and the broader ecosystem has raised concern and generated significant scientific literature. There has been extensive literature on the issue of ecological and genetic effects from hatchery releases on wild populations, but quantitative research and monitoring of these impacts are generally lacking (Laikre *et al.* 2010). Studies measuring impacts on wild conspecifics and other competitive species are also sparse in the stocking literature.

The environmental impacts of stocking fish are generally negative as a result of predation, competition, habitat alterations, disease, and the loss of genetic integrity. A brief overview of these impacts is provided in this section. For more detailed information we refer you to the numerous reviews on the biological impacts from stocking fish that are available in the literature (*e.g.* Arthington 1991, Cowx 1998, Lorenzen *et al.* 2001, Brown and Day 2002, Welcomme and Vidthayanon 2003, Cowx and Gerdeaux 2004, De Silva and Funge-Smith 2005, Bell *et al.* 2006, Vitule *et al.* 2009, Russell *et al.* 2013, Ingram and De Silva 2015, Taylor *et al.* 2017).

3.4.1.1 Genetic impacts

Stock enhancement programs have the potential to affect the genetic diversity of wild populations (Allendorf 1991, Cooke *et al.* 2001, Aphrahamian *et al.* 2003, Hara *et al.* 2008, Araki and Schmid 2010). The perceived importance of genetic impacts on wild populations is highlighted by the sheer volume of published literature on the topic, including numerous reviews (*e.g.* Allendorf 1991, Hindar *et al.* 1991, Waples 1991, Busack and Currens 1995, Campton 1995, New South Wales Fisheries 2003, Utter 2003, Araki and Schmid 2010). Refer to those reviews and others for more details of the topics outlined below.

Stocking programs have received substantial criticism based on both real and perceived impact of hatchery-bred fish with altered or inferior genetic make-up breeding with wild populations, resulting in loss of genetic diversity or loss of viability (Ryman 1981, Allendorf 1991, Meffe 1992, Philipp *et al.* 1993, Brown and Day 2002). Genetic diversity is positively correlated with fitness (Reed and Frankham 2001) and is critical for the long-term survival of populations, providing adaptive potential to cope with environmental change, new diseases, parasites, predators and competitors (Soule 1990, Hilborn *et al.* 2003, Schindler *et al.* 2010). The major risk is if the level of within and among-population genetic diversity is lowered significantly, thereby reducing the genetically effective population size and fitness (Lorenzen *et al.* 2010).

Cooke *et al.* (2001) argued that although stocking usually occurs to supplement natural stocks and increase abundance, unless genetic integrity is maintained, it is most likely the enhancement would be counter-productive. Genetic changes are often more difficult to document and monitor than demographic or ecological effects (Allendorf 1991). These impacts have received a lot of attention, but the literature is mainly theoretical in nature (Keenan 2000). Stocking can influence wild fish populations in several ways and the rapid development of genetics technologies for studying the genetic structure of populations has shed considerable light on how stocking activities have affected species and populations that are the subject of stocking programs (*e.g.* Nguyen *et al.* 2015).

Changes in genetics caused by inbreeding and outbreeding depression, and adaption to captivity, can lead to reduced fitness in the wild and major long-term disadvantages in the natural environment (Frankham 1999). Interbreeding may result in negative, genetic consequences directly or indirectly through changes in population size, pathogens and parasites, predation, competition, etc. (*e.g.* Hindar *et al.* 1991, Carvalho 1993, Skaala *et al.* 2006). In the review of 21 studies comparing hatchery and wild stocks, Araki and Schmid (2010) found 12 studies reported negative effects of hatchery rearing on the fitness, eight studies suggested lower reproductive success, and four studies suggested lower survival rates of hatchery fish. Notably, however, six recent studies addressed fitness effects but did not find evidence of the negative effects, indicating that fitness effects can be small at least in some cases.

Genetic effects can occur when interbreeding between released and native fish transfers genes from released fish into the native population (Utter 2003). Inbreeding is the mating of closely related individuals that share common alleles by decent and leads to increased homozygosity in a population and is problematic when hatchery genotypes lead to cross-bred offspring that are less well adapted for survival in the wild (Moore 2000). Conversely, outbreeding depression is defined as the erosion of population fitness through mating of genetically divergent populations (Waples 1991). Persistent release of individuals into the wild from a single source can homogenize genetic variability among populations. This can result in a decline in overall or reproductive fitness, but this is less common than inbreeding depression (Ralls *et al.* 2013).
Using a limited number of founding broodstock for stocking programs can create a genetic bottleneck, which can lead to the loss of genetic diversity in a wild population. The potential reduction in effective size of a wild population resulting from a small genetic effective size of hatchery releases is termed the Ryman-Laikre effect (Ryman and Laikre 1991, Tringali and Bert 1998). Much of the evidence supporting these views is based on studies of northern hemisphere species, particularly salmonids

(Araki and Schmid 2010). Hybridizations between hatchery and wild individuals have the potential to lower the fitness of the wild population (Kostow 2004, Araki *et al.* 2007, 2008). For example, a 25% reduction in heterozygosity in Rainbow trout (*Onchorynchus mykiss*) as a result of inbreeding, was shown to decrease fry survival by 19%, growth by 23% and increase phenotypic deformities by 38% (Kincaid 1975 a,b).

Several empirical studies show evidence of substantial gene flow from hatcheries and changes in genetic compositions in wild populations (Araki and Schmid 2010). However, fitness reduction in stocked populations caused by genetic effects of captive rearing has not been widely reported, and this might be attributed to low strength in the methods used and lack of data. For example, a review of genetic effects of hatchery fish on Pacific salmon and steelhead found only limited empirical data demonstrating such effects on wild populations (Campton 1995). Conversely, Kitada *et al.* (2019) found that the stock enhancement programs for Red sea bream reduced genetic diversity of the populations, but the genetic effect diminished with increased size of the wild population. The survival rate reduced by 49% per generation, but the fitness of hatchery fish might not be reduced until 3.5 generations in captive rearing. The rate of fitness reduction in a hatchery-reared population was cohort specific, but exponentially decreased as time duration in captivity increased. Similar results were reported in a meta-analysis of Steelhead, Brown trout and Atlantic salmon, which showed captive breeding reduced relative reproductive success by ~40% per generation in captivity (Araki *et al.* 2007).

Estimations of the genetic impacts of fish stocking in Australia are rare, but this activity is often classed as a threat to wild native fish populations (*e.g.* Koehn 2005, Lintermans *et al.* 2005). The results from published studies are mixed, with no clear consensus. In the Murray-Darling Basin, Gillanders *et al.* (2006) claimed that the many hatchery-produced fish stocked have reduced genetic diversity and reduced fitness. However, Rourke *et al.* (2010) found that there have been no major temporal changes in genetic diversity, heterozygosity, allelic richness and effective population size of pre- and post-stocking Murray cod in the southern parts of the Murray-Darling Basin and suggested that the use and regular replacement of wild-caught broodfish, and mixing progeny from different spawnings before stocking contributed to the lack of genetic change.

Similarly, in northern Australian, Russell *et al.* (2013) discovered that despite many years of stocking Barramundi into the Johnstone River, there was no evidence for a loss of genetic diversity, or introgression of genes from the original broodstock back into the wild population. Conversely, Leahy *et al.* (2022) reported Barramundi stocked into an upstream impoundment represented only 3% of the commercial barramundi fishery in the Dry Tropics, but hatchery ancestry was detected in 21% of the catch. This indicated that stocked fish successfully breed with wild fish and contributed genetic material to subsequent generations. The strong representation of hatchery ancestry among the wild-born population highlights the importance of fish stocking regulations to support local genetic diversity and evolutionary traits.

In other freshwater systems, a significant loss of heterozygosity and allelic richness in Eastern freshwater cod since stocking has been identified (Nock *et al.* 2011). This loss of genetic diversity may be due to the use of insufficient broodfish and low effective population size at the hatchery (Rowland 1990), and/or swamping of the lower Mann–Nymboida population by hatchery-reared fingerlings following some stocking events (Nock *et al.* 2011). Nguyen and Ingram (2012) also reported reduced genetic diversity of stocked Chinook salmon in Australia, but there was no evidence of genetic bottlenecks or inbreeding.

In estuarine waters, the prevalence of inbreeding was found to be not demonstrably greater among the restocked than wild Black bream *Acanthopagrus butcheri* in the Blackwood River in Western Australia. Thus, any discrepancy in the biological performance of restocked versus wild fish, such as the slightly lower growth rate of the former, is likely to be driven by environmental rather than genetic factors (Gardner *et al.* 2013). The genetic data indicated, however, that some of the rare alleles in wild fish were absent in restocked fish (Gardner *et al.* 2013). Such a loss of alleles is relevant from a management perspective because it could potentially diminish the evolutionary potential of the population (Willi *et al.* 2006).

3.4.1.2 Minimising genetic impacts by building on hatchery practices

For production of animals that are to be stocked into the wild for fishery enhancement programs, it is important to breed from wild caught broodstock and not from animals that have been held in captivity over multiple generations (Kitada 2020). If inbred hatchery produced animals are stocked where they outnumber wild stock this can lead to loss of genetic diversity in the wild (Ryman and Laikre 1991). Crossing domesticated stock with wild caught stock prior to stocking can improve outcomes. For example, domesticated White spotted charr *Salvelinus leucomaenis* females crossed with wild males had offspring that survived in the wild 2.5 times better than charr produced from domesticated stock alone (Yamashita *et al.* 2020).

As a method to reduce genetic risks, the feasibility and genetic implications of using wild-caught fertilised eggs to culture a marine fish for restocking or stock enhancement purposes has received comparatively little attention. Restocking or stock enhancement programs have traditionally relied on captive broodstock to produce individuals for release into the wild. Greater genetic diversity of released individuals presumably could be assured by culturing fertilised eggs/larvae collected in the wild rather than culturing the offspring of hatchery-held broodstock (Munro and Bell 1997, Crossman *et al.* 2011). Fish species which form spawning aggregations, such as epinephelids, sciaenids, labrids, lutjanids and sparids may provide the opportunity for wild egg collection, especially where large numbers of fertilised eggs can be relatively easily collected (Heyman *et al.* 2001, Wakefield 2010, Bowling 2014, Partridge *et al.* 2016). Similarly, invertebrates and shellfish species which are broadcast spawners with a settling phase (*e.g.* scallops and oysters), also have the potential for wild juvenile collection (Drummond *et al.* 2014). If significantly higher survivorship of wild-collected individuals to an early juvenile stage can be achieved in cultured environments, this approach should enable supplementation of wild stocks upon the release of the reared individuals.

Partridge *et al.* (2016) demonstrated that wild spawned snapper eggs can be captured from spawning aggregations and cultured in high numbers. Several orders of magnitude better survival of fertilised eggs, larval and juveniles could be achieved by hatchery rearing wild-collected Snapper (*Chrysophrys auratus*) eggs, when compared to their survival in the wild, with no loss of genetic diversity compared to the wild population (Prokop 2016). Similar results were also observed by Drummond (2004) in the NZ scallop fishery. The successful collection and rearing of wild-collected spat were a key component of their highly successful scallop enhancement program.

This technique also has the potential to greatly increase the cost efficiency of producing juveniles for release. Capturing, maintaining and spawning broodstock in a hatchery comes at a significant cost

(Sato *et al.* 2014). Large numbers of broodstock are typically needed to ensure the genetic composition of the released fish are representative of the wild population (Blankenship and Leber 1995, Lorenzen *et al.* 2010). The collection of wild eggs or spat eliminates the costs associated with broodstock and spawning, greatly reducing the overall costs for producing stock for release (Partridge *et al.* 2016). However, the consistency of wild egg or spat supply is a potential risk in using this approach, especially if wild spawning aggregations and egg collection are highly dependent upon highly specific environmental conditions.

If not collecting wild spawned seed, it is important to hold and turnover sufficient numbers of wild broodstock to maintain genetic diversity and adaptive potential (Grant *et al.* 2017). When wild-sourced stock are used for broodstock each generation, the introduction of new genetic material reduces loss of genetic diversity (Mobrand *et al.* 2005, Grant 2017). Broodstock should be sourced from populations with individuals that are genetically compatible with individuals at the release site as there may be adaptations to local environmental conditions that will influence survival (Grant *et al.* 2017). For example, there were problems with the survival of hatchery-reared Roe's abalone derived from broodstock sourced in southern Western Australia, when stocked onto a reef near Kalbarri, (central west of Western Australia) where water temperatures are warmer than what would be experienced by southern stock (Strain *et al.* 2019). Using the correct genetic stock and maintaining high N_e should also be followed when stocking into impoundments. In impoundments, the stocked fish are generally unable to breed, but in wet years there is always the possibility of diadromous species passing safely over a spillway and mixing with breeding stock downstream in estuarine areas, or potadromous species migrating upstream and downstream from an impoundment to mix and breed with wild stock in riverine areas. Hatchery practices should be optimized to increase the effective number of captive breeders and thus also the total effective population size (N_e) in the population (Hagen *et al.* 2020).

Using the principles above, NSW Department of Primary Industries (Fisheries) have developed guidelines for hatcheries producing animals for freshwater and estuarine enhancement. The guidelines in both documents are similar. The freshwater document (NSW Department of Primary Industries 2019) breaks up the state into geographical genetic zones for each of the species used for stock enhancement. This ensures use of genetically compatible stock for any region proposed to be stocked. The hatchery must have a minimum of ten pairs of broodstock for each broodstock genetic region. For each stocking run the hatchery must attempt to spawn eight separate male-female matings. From 1 to 3 males may be used in each female/male matings. For each production run for stocking, the hatchery must use the progeny of at least five separate female/male matings. The matings from each production run must occur within a four-week period. The progeny of each mating must be kept separate during spawning and incubation, and equal numbers of larvae from each successful spawning (minimum of five spawnings) must be pooled to make up a batch prior to stocking a fingerling pond. Each broodfish should not be held for more than five breeding seasons. All broodfish should be individually identifiable (tagged) to ensure management of matings. These measures ensure reasonably diverse progeny are stocked each production run, and regular turnover of broodstock with new wild sourced fish helps prevent domestication and helps maintain a high N_e over several years of stocking (Blount *et al.* 2017). The marine document is almost identical in application. There are genetic geographical zones for each species used for marine/estuarine stock enhancement and there is a requirement to use wild caught broodstock. Hatcheries must produce fish using similar protocols to the freshwater hatchery quality assurance guidelines to achieve an N_e of 50 (NSW Department of Primary Industries 2014). This can be achieved by using five different pairs of broodstock each year to contribute to a stocking event, with the broodfish changing each season over a five-year period (Blount *et al.* 2017).

The genetic objectives of stocking should be to maximise the effectiveness of the program while minimising detrimental effects on natural populations or species (Cross 2000). To conserve the genetic diversity of wild stocks, broodstock selection and management is a key consideration. Strict protocols governing broodstock management, hatchery procedures and stocking strategies can and should be implemented to prevent translocation of exogenous genes and minimise changes in allele frequencies of wild stocks (Farrington *et al.* 2004, de Innocentiis *et al.* 2008). Given the continued increase in stocking of hatchery-reared fish and the potential for interactions with wild fish, it is essential to take a responsible approach and to monitor and experimentally evaluate any stocking program.

3.4.1.3 Ecological impacts and carrying capacity

The ecological risks of stocked animals from hatcheries, especially competition for limited foraging resources and increasing predation on natural populations, have been recognized for some time (Anderson *et al.* 2020). It is well established that fish have a strong influence on ecosystem processes and the mechanisms by which this occurs are complex and diverse (Arnason 2001, Achord *et al.* 2003, Pearsons 2008). Ecological impacts can occur when hatchery fish negatively affect how wild fish interact with each other, their environment, and/or other species (Kostow 2009, Lorenzen *et al.* 2010). Numerous reviews have been undertaken on the effects of stocking practices on the receiving environment and endemic species (*e.g.* Arthington 1991, Lorenzen *et al.* 2001, Brown and Day 2002, Welcomme and Vidthayanon 2003, Cowx and Gerdeaux 2004, De Silva and Funge-Smith 2005, Bell *et al.* 2006, Vitule *et al.* 2009). These reviews have identified competition and predation as the main drivers behind potential ecosystem impacts from stocking fish.

Although stocking programs try to optimise social and economic benefits and to foster development of the various fishing sectors, their primary consideration must be sustainability of the fish resources and their environment (Borg 2004). Employing stocking as a tool in fisheries management requires a thorough understanding of the ecological processes that provide the potential for stocking within different ecosystems. Most stocking projects involve the release of species native to the area. The movement of fish species beyond their natural range is strongly discouraged as it is potentially one of the most ecologically damaging of human activities whose effects can be difficult to reverse (Koehn 2004).

Crucial to any form of fishery manipulation is understanding whether or not the ecosystem can sustain and support the released fish. Carrying capacity is defined as the general productivity of a given region and includes food, habitat, shelter, predators and competitors (Kashiwai 1995). It is important to consider the carrying capacity for the fish of the size being released and larger sizes, as their requirements will change as they develop. Exceeding the carrying capacity of a system can result in trophic cascades, poor survival and even cause extinctions of wild species (Arthington 1991). A decline in the density and abundance of one species can result in one or more other species increasing in volume because of spare capacity in the system.

Carrying capacity can vary greatly between different systems and environments. For example, the growth rates of Barramundi stocked in impoundments can be approximately three times that of fish stocked in rivers due to the carrying capacity and food resources in the highly productive impoundment waters (Russell and Rimmer 1997, Rimmer and Russell 1998). In most cases, stock enhancement is considered for fishery stocks that have already experienced a significant decline. Therefore, the carrying capacity for the target species may already have been altered by the fishing pressure. Vacant niches initially created by fisheries extraction could have been filled by other species, thus varying the carrying capacity for the stocking species. It is critical to clearly understand the carrying capacity of the receiving environment at the time of stocking to determine the likely levels of release that can be supported without detrimental population or ecosystem effects.

Competitive interactions between fish and their resulting impacts can have overarching negative effects on fish abundance and behaviour (Gillanders *et al.* 2006). Changes in abundance and behaviour primarily arise through competitive interactions between stocked and wild fish. Changes due to competition can be either direct (for food and habitats, Begon *et al.* 1996, Peery *et al.* 2004) or indirect (habitat alteration, behavioural changes, expansion of range, displacement of wild stocks, Fletcher *et al.* 1985, Gillanders *et al.* 2006).

Competition largely underpins the ecological mechanisms behind the interactions between stocked fish and wild fish populations. The complexity of ecosystems means that stocking fish can add additional pressure to existing resources which could alter community structure or the trophic food web (Ingram and DeSilva 2015). Density-dependent competition for resources such as food and space is a primary mechanism through which this can occur (Pearsons 2008).

Competition generally evokes negative effects to either stocked fish or wild fish populations through reduced growth, survivorship and spawning success, changes in resource use and displacement of stocks (Lachance and Magnan 1990, Fjellheim *et al.* 1995, Weiss and Schmutz 1999, Imre *et al.* 2005). For example, in a review by Einum and Fleming (2001), fifteen of sixteen studies reported reduced survivorship of stocked fish, compared to their wild counterparts.

The impact of stocked fish on food resources is likely to depend on available resources, size of fish stocked and the adaptive abilities of stocked fish (Sayer *et al. 2019*). The competitive influences of stocked fish on wild populations are largely density-dependent and will be greatest where resources are limiting (Ochwada-Doyle *et al.* 2012). The displacement of wild individuals of both target and non-target species, can occur with persistent releases of hatchery-reared offspring when release numbers are large, and habitats are near carrying capacity (Tringali and Bert 1998). If hatchery fish compete with wild fish in an environment with limited carrying capacity, hatchery fish may replace rather than augment wild populations (Hilborn 1992, Hilborn and Eggers 2000). Density-dependent factors can intensify competition for food or space and can alter growth patterns and reproductive output, which in turn can reduce the effective population sizes and potentially lead to the loss of adaptive potential (Ochwada-Doyle *et al.* 2012).

Attention should be paid to the influence of stock enhancement on other species, especially when releasing carnivorous species. Positive and negative predation effects can occur. Stocked fish may act as prey for the existing fish in the system, especially in their juvenile stages, subsequently providing a positive benefit for the wild population and a negative effect for the stocked fish (SKM 2008). Conversely stocked fish may prey upon wild fish and therefore negatively impact wild fish whilst improving the survival of the stocked fish. For example, Sudo *et al.* (1992) reported that released juvenile flounder were eating considerable numbers of natural juvenile red sea bream.

Predation by stocked fish has been associated with the decline of several native fish species. Impacts from predation can occur through increased mortality of wild fish from direct consumption (McDowall

2006), or from changes to trophic structure of aquatic ecosystems through alteration of the food chain (Leberer and Nelson, 2001, Eby *et al.* 2006, Hebert *et al.* 2008). For example, in New Zealand, introduced salmonids have been demonstrated to not only predate upon small native fish species, but also consume the majority of all benthic invertebrate production, altering the composition of the aquatic community and available food resources for other species (Huryn 1998).

Predation on smaller native species by released fish can be particularly pronounced when novel non-indigenous species are stocked into naïve systems (Koehn 2004). The scale of impacts from stocking novel species can be quite large. One of the classic examples in this regard is the introduction of Nile perch (Lates niloticus) into Lake Victoria in the 1950s, which contributed to the extinction of up to 260 endemic fish species (Leveque 1995). Another example is the introduction of grass carp into Donghu Lake, China, which resulted in the decimation of submerged macrophytes. The subsequent ecological changes brought about an upsurge of bighead carp and silver carp populations and the disappearance of most of the 60 fish species native to the lake (Chen 1989).

Case study 2: The introduction of Nile perch into Lake Victoria

One of the clearest examples of the conflict between economic benefits and ecological consequences created by stock enhancement is the introduction of the non-native Nile perch (*Lates niloticus*), into Lake Victoria, Africa's largest lake. The piscivorous Nile perch was introduced to create higher-value commercial and sports fisheries Beadle (1981). Before the introduction of Nile perch, the fish fauna of the lake was dominated by more than 500 endemic haplochromine cichlid species, which formed the basis of the local fishery (Witte *et al.* 2007). The growth of the population in Lake Victoria was very rapid and within 25 years of its introduction the Nile perch became ubiquitous, occurring in virtually every habitat with the exception of swamps and affluent rivers. Since introduction, it has completely transformed the fishing industry and the species composition of the fish fauna of the lake. Nile perch has preyed on all other species with profound effects, especially on the stocks of cichlids. These originally comprised 80% of the total fish biomass in Lake Victoria, but decreased to less than 1% of fish catch (Achieng 1990). This impact was accompanied by a series of ecosystem changes and helped Nile perch be listed as one of the 100 worst invasive species (Lowe *et al.* 2000). The original multispecies fishery changed dramatically to one based on only three species, with Nile perch contributing up to 60% of the total annual fish catch (Achieng 1990).

The introduction of Nile perch also had significant socio-economic impacts. Locals experienced a sharp decline in the catch of the traditionally more desirable fish species. The more limited range of fish available for their consumption initially put substantial pressure on the artisanal fishing industry. Local fishers eventually switched to targeting Nile perch and the economic benefits quickly became apparent, with a dramatic increase in fish production and the development of an overseas export market (Aloo *et al.* 2017). The number of boats operating in the fishery increased six-fold and Nile perch has since become the most important commercial catch, supporting a major and thriving industry on a scale not anticipated either by those who introduced Nile perch or by those who opposed its introduction into the lake.

Compared with the global published literature, there is limited information on the ecological impacts of enhancement of natural stocks on wild conspecifics or the receiving ecosystems in Australia. The severity of ecological impacts from fish stocking appears closely linked to the type of species stocked and how novel they are to the receiving ecosystem.

Stock enhancement of fish into an existing wild population may only have minimal ecological effects. Russell *et al.* (2013) assessed barramundi stocking in the Johnstone River and Tinaroo Falls Dam and concluded that no significant demonstrable ecological impacts could be attributed as stocked fish did not naturally move into areas that are outside the normal range of wild fish, no difference was found in the diets of stocked and wild fish, nor was there evidence to suggest that stocked fish preyed upon species of conservation concern.
Similarly, ecological change following enhancement of lobsters within their natural range in both South Australia and Tasmania resulted in no significant changes in either predator or prey abundances (Green *et al.* 2012). Other studies have shown that reduction in density of lobsters through fishing can have ecological impacts such as release of predation pressure on urchins and also emergence of abalone (Pederson and Johnson 2006, Pederson *et al.* 2008). Restoring lobster density to more natural levels through enhancement would thus be expected to beneficial by compensating for any ecosystem effects of fishing.

Translocation of native species to new areas is likely to have more detrimental ecosystem effects on the receiving environment. The introduction of several native, but non-endemic predatory fish species into Lake Eacham was believed to have led to the localised extinction of the Lake Eacham rainbowfish due to their limited distribution (Barlow *et al.* 1987). A further example was the translocation for fishery purposes, of the large piscivorous Sleepy cod *Oxyeleotris lineolatus* into the upper reaches of the Burdekin River, Queensland, where the species did not naturally occur. The release and spread of this species is believed to have driven a significant decline in abundance of the purple-spotted gudgeon *Morgunda adspersa* in the region (Pusey *et al.* 2006).

Evidence from introducing new exotic fish species, particularly for recreational fishery development, can have significant ecological and conservation impacts. The decline of several native galaxiid species in alpine systems has been attributed to predation by non-native salmonid species introduced for recreational fisheries in Australia and New Zealand (Tilzey 1976, Pearsons and Fritts 1999, Lintermans 2000, McDowall 2006). These impacts were exacerbated by the broad dietary overlap between the salmonids and galaxiids, which also limited food resource availability for the native galaxiids (McDowall 2003). Redfin perch are another species whose introduction through stocking has had negative impacts through predation of small native fish and fry (Thorn 1995, Morgan *et al.* 2002).

3.4.1.4 Impacts on wild populations of the stocked species

Stocking enhancement or translocation can have significant impacts on the wild populations of the released species (Hilborn and Eggers 2000, Bohlin *et al.* 2002, Camp *et al.* 2014). The goal of many enhancement programs is to increase the standing stock of a species, but this does not always occur. Stocked fish can make a substantial contribution to wild populations and enhance stock abundance, particularly where natural recruitment is poor or highly variable (Taylor *et al.* 2009, Crook *et al.* 2010, Crook *et al.* 2016). For example, stocking activities have contributed to an increase in the distribution and abundance of Murray cod in some areas and may have been responsible for their recovery in the Gwydir River catchment (Rowland 2005). However, this has not occurred everywhere and there has been no evidence of survival of stocked Murray cod in the Lachlan River catchment, suggesting that recovery of this population in recent decades has been due to natural recruitment alone (Rowland 2005, Rourke *et al.* 2011).
However, one of the most difficult issues to resolve is if stocking is actually increasing production or simply displacing existing wild stocks through density dependent processes (Welcomme and Bartley 1998, Lorenzen *et al.* 2010). Releasing hatchery-reared fish into wild populations can be problematic

because hatchery fish are often stocked on top of the natural production, which has become constrained by habitat loss (*i.e.* reduced natural carrying capacity), thus inducing potentially deleterious competition between the wild and released fish (Einum and Fleming 2001). Replacement of wild fish by hatchery-reared fish and their offspring may take place with or without any associated increase in total population abundance (Rogers *et al.* 2010). There have been many examples of stock enhancement where the released fish have replaced or dominated wild fish populations (*e.g.* Welcomme and Bartley 1998, Whittier and Kinkaid 1999, Einum and Fleming 2001, Goldburg *et al.* 2001, Loneragan *et al.* 2004, Hamasaki and Kitada 2006, Brennan *et al.* 2008,). Growth and reproduction can be impacted after translocation or the release of hatchery-reared fish (Svåsand *et al* 2000, Chandrapavan *et al.* 2010, Green *et al.* 2010, Gardner *et al.* 2103). Biological interactions between hatchery-reared and wild fish may also result in a reduction of the abundance of fish with wild characteristics, even when overall abundance of fish, catches and fishing effort are increased by the enhancement (Camp *et al.* 2014). Differences between hatchery-reared and wild fish are at least in part genetically based and replacement may therefore persist for multiple generations (Lorenzen 2005, Quinn *et al.* 2006). Conspecific predation from releasing fish does not always impact the population structure of a species. Russell *et al.* (2013) found no evidence to suggest widespread cannibalism was impacting the contribution of stocked and wild barramundi to the population structure in the Johnstone River or Tinaroo Falls Dam.

The biotic and abiotic factors contributing to the carrying capacity of a system are typically highly variable but very important for determining the stocking magnitude that will result in an increase in the population rather than displacement of the wild stock (Støttrup and Sparrevohn 2007). This is rarely estimated in stocking programs, and may be difficult to predict in cases where stocking aims to fill the carrying capacity when supply of juveniles fails. Understanding the population dynamics of a fishery is a critical component of successful stock enhancement programs (*e.g.* Caddy and Defeo 2003, Lorenzen 2005, 2008, Ye *et al.* 2005, Leber 2013). Decisions about stocking magnitude become clearer when density, biomass, and distribution of the target population are monitored both prior to and following hatchery releases (Brennan *et al.* 2008, Becker *et al.* 2021). Successive monitoring of biomass and stocking can help reveal the productive capacity of the release site.

3.4.1.5 Disease and pathogens

Stocking of fish can lead to the transmission or introduction of infectious diseases and pathogens which can have severe impacts on wild fish populations (Gaughan 2002, Peeler and Murray 2004, Naish *et al.* 2008). The introduction of pathogens via stocking is a problem throughout the world (Paperna 1991). For example, the stocking of redfin perch in Australia resulted in the spread of a dangerous virus to trout and several native fish (Thorn 1995). An organism exposed to a new disease or pathogen may not necessarily die from becoming infected, but the resulting infection can negatively influence immunity, growth, feeding ability, reproduction ability and distribution (Cunningham 1996).

Disease introduction can also impact the socio-economic value of stocked fisheries. For example, the monogenean parasite *Gyrodactylus salaras* caused losses to Atlantic salmon fishing industries in Norway following its introduction from infected hatcheries through fishery enhancement programs (Johnsen and Jensen 1991).

High densities of aquatic animals in a hatchery setting significantly increases the risk for diseases and parasites to take hold and multiply. Although some hatchery disease and parasite outbreaks may be diseases already endemic in the wild, stocking of diseased or parasite ridden animals may compromise their post-release survival. Management of disease risks is an essential component of good stocking practices (Lorenzen *et al.* 2010) and the use of risk analysis frameworks can greatly reduce the chances of spreading diseases resulting from releasing cultured juveniles (Bartley *et al.* 2006). Good hatchery and health management practices and adherence to criteria in quality assurance will eliminate or minimise the transfer of pathogens from hatcheries (Rowland 2013). Various hatchery codes of practice have been developed and all contain components designed to minimise disease and parasite risk and accidental translocation of unwanted organisms.

The Aquaculture Association of Queensland (AAQ) (2007) voluntary code of practice recommends a hatchery health management plan. This includes quarantine arrangements, sending any fingerling abnormalities for diagnostic investigation, testing of Barramundi *L. calcarifer* for Nodavirus at 21 days and 42 days post-hatch, record keeping of any treatments of broodstock and fingerlings for protozoan ectoparasites and internal worms, and an ability for the hatchery to demonstrate the results of their structured program of disease monitoring, including health testing and monitoring of broodstock and their progeny. There is also a requirement for stocking groups purchasing from AAQ hatcheries to complete a check list of fingerling condition before acceptance of the delivery. AAQ also have an objective to eliminate all insect, plant and non-target finfish species from fingerling consignments but do not actually outline how to achieve this. Some options could include screening of water intakes (to eliminate non-target species of fish from entering grow-out areas) and rigorous checking of consignments of fish held in tanks prior to transport to stocking sites to eliminate things such as aquatic plants, insects and tadpoles that may not be prevented from entering nursery ponds by intake screens.

The New South Wales Hatchery Quality Assurance Scheme (NSW Department of Primary Industries 2019) is more rigorous than the AAQ code of practice and would appear to be a better model to follow. It includes a requirement for a written Health Management Plan (HMP). The plan must include a disease surveillance routine. The hatchery must have a sterilisation procedure for nets, buckets and other equipment. The hatchery must have a binocular or monocular microscope with a powered light source (for examining specimens) and must also have sampling, dissection and specimen submission equipment, and all new broodstock entering the hatchery must be quarantined. Hatcheries must also complete a Dispatch and Health Statement for each consignment of fish destined for stocking programs. Ten fish from each batch must be examined less than 24 hours prior to consignment and the hatchery must not dispatch a consignment having detected non-target species, including, insects, snails, tadpoles, vegetation, moribund fish, and fish with signs of disease. Each consignment must be quarantined for a minimum of 24 hours post-harvest before shipping off-site. Additionally, the hatchery must use a saltwater bath prior to the dispatch of any fish destined to freshwater sites, or a freshwater bath if fish are to be released into saltwater. Records of disease testing must also be retained. Marine species approved for stocking in New South Wales must also comply with the NSW Hatchery Quality Assurance Scheme (NSW Department of Primary Industries 2014).

The protocols in the NSW Hatchery Quality Assurance Scheme are common sense and should help protect the receiving environment and ensure that the fingerlings arrive in the best possible condition, provided hatcheries comply with the recommendations (regulations). Similar practices should be adopted by hatcheries Australia-wide. Compliance to such codes of practice or quality assurance schemes is essential to ensure healthy, non-contaminated batches of fish are stocked.

3.4.2 Socio-economic impacts

Success in fisheries management is often measured against an increasingly broad set of criteria: biological (yield, ecosystem indicators), economic, social, and institutional attributes (Charles 2001, Garcia and Charles 2007). However, a fundamental consideration is whether a stocking program would yield a positive economic outcome (Lorenzen 2008). The economic feasibility of stock enhancement is a cost-benefit problem where benefits of increased catch, higher value or lower fishing costs, need to exceed the costs of rearing and releasing fish (Tlusty 2004). For a stocking action to be considered economically attractive, the cost of juvenile production must be as low as possible without compromising the quality of the released fish, and the catch per unit effort must also increase significantly (Lee 1994, Moksness *et al.* 1998, Borthen *et al.* 1999).

Stocking programs have demonstrated that in some cases it is possible to increase fishery landings (*e.g.* Drummond 2004, Hart and Strain 2016), but economic feasibility, in both recreational and commercial fisheries, depends on many factors. Four key drivers behind the economic profitability are:

1. cost to produce and stock fish
2. recapture rate (survival to harvestable size and contribution to the fishery)
3. market value of recaptured fish (landing value or the value of angler experience)
4. the cost of negative impacts (ecological, genetic, economic and social)

These topics have formed the focus for much of the research into improving the economic feasibility of releasing fish to enhance fisheries. Stock enhancements not only need to be economically viable, but they need to add value to, or outperform alternative management measures such as fisheries regulation or habitat restoration, which are often either cheaper or provide a wider range of benefits. Bartley and Casal (1998) reported that most of the recorded ecological effects of introducing species are negative; however, reported socio-economic impacts were mostly positive and often outweighed the negative ecological impacts. Although it may not be possible to put dollar values on all these costs and benefits, they do need to be recognised and attempts made to value them. These considerations highlight why evaluating the effectiveness of stock enhancement projects is recognised as being fundamental to justifying their expense (Rutledge and Matlock 1986, Blankenship and Leber 1995, Cowx 1998, Lorenzen *et al.* 2010).

In economic analysis, the quality of input determines the quality of the output. Stocking programs historically were rarely accompanied by appropriate cost-benefit assessment, and where conducted, the level of economic scrutiny has varied so greatly, that direct comparisons can often be difficult, if not impossible. A variety of methods have been used to evaluate the economic feasibility and socio-economic effectiveness of stocking (Lorenzen *et al.* 2010). Unfortunately, robust, comprehensive cost benefit analyses or bio-economic modelling was rarely applied until recently. In the past, comparisons were made between catches before and after the stocking, but this approach was unreliable since any positive effect might be masked by the natural inter-annual variability of the stock's abundance. Costs for raising and releasing fish were usually identifiable, however, unless released stock could be accurately identified and monitored, yield or benefit per released fish were not available (Blaxter 2000).

Development of viable mass marking of released fish to discriminate hatchery-reared fish from wild stock, enabled a greater range of economic assessment approaches to be utilised. Several simplified cost-benefit indices were developed to compare the economic feasibility between different stocking release strategies, species and release sites. Many studies determined the success of a stocking program by assessing either post-release survival, contribution to the receiving population, or the yield or percentage of stocked fish in the total catch (Evans and Willox 1991, Vollestad and Hesthagen 2001). These measures enable comparison between the costs of producing fish for release and benefits derived from their capture. The cost of released fish can be summarised in three ways: cost per fish released, cost per adult survival, and cost per adult harvested (Naish *et al.* 2007).

Cost-benefit analysis for enhancement is rarely straightforward, especially in open systems, since benefits can be difficult and expensive to identify (Blaxter 2000). Consequently, cost-effectiveness analyses have ranged from basic comparisons of production costs to commercial landed value (*e.g.* Kitada *et al.* 2018), through to comprehensive benefit-cost analyses and economic impact assessments (*e.g.* Hunt *et al.* 2017). Many comparison techniques vary in the extent of the data they incorporate. For example, measuring the market value of each caught fish, cost to produce each stocked fish and number required to survive until capture to ‘break even’, is a commonly applied method for evaluating the cost-effectiveness of fish stocking, without including the total benefits of the stocked fishery (Loomis and Fix 1999, Brown and Day 2002, Aphrahamian *et al.* 2003). Similarly, economic efficiency has been defined as the ratio of net income from commercial landed value to the cost of producing the fish for release (Kitada *et al.* 1992, Kitada 2018, 2020). Economic efficiency is often used as the metric of the economic performance instead of NPV because various data, such as annual costs for harvest, management and interest rates, are typically not available in many instances (Kitada 2019). If the NPV method were used, then the economic efficiency estimates obtained here would be less (Kitada 2018). These relatively simplistic evaluations enable cost-effective comparison between different stocking programs or strategies, but cannot easily be compared to other management options.

Development of supply chains for domestic and overseas high-value live markets requires consistent supply. Climatic variability and highly variable natural recruitment can generate inconsistent catches (Loneragan *et al.* 2004). Re-stocking and stock enhancement following recruitment failure could lead to faster fishery recoveries, and also be used to supplement natural recruitment to provide a more consistent and higher yield harvest from year to year. Pinkerton (1994), for example, describes economic benefits resulting from Alaskan Salmon *Oncorhynchus spp*. enhancements that result from greater consistency and quality of harvests, as well as greater volume.

Measuring the impacts on the value of recreational fishing from any stock enhancement program is considerably more challenging, as it is frequently the activity itself rather than the outcome of the activity (the catch) that is valued by the fishers (Camp *et al.* 2015). Valuing the harvest caught by recreational fisheries, as done in commercial fisheries, would considerably underestimate the value attributed to the activity by those fishers who are likely to fish for reasons independent of numbers or species caught. Instead, non-market valuations, such as revealed or stated preference methods, are required to obtain an accurate value and evaluate the marginal impacts of stocking (Whitmarsh and Pickering 2000). As Tietenberg and Henk (2004) noted, intangible benefits (such as non-use values in this case) should be quantified to their fullest extent, else the net benefit estimates will be downward biased.

Numerous studies have investigated marginal increases in angler satisfaction from additional catch (*e.g.* Cox and Walters 2002, Cox *et al.* 2003, Hunt 2005, Arlinghaus *et al.* 2014, Beardmore *et al.* 2015, Camp *et al.* 2015), but these have rarely been derived regarding stocking. However, in some circumstances the results can be combined with changes in catch rate data to estimate the marginal recreational benefit from stocking activities. As satisfaction and economic activity are key benefits derived from recreational fishing (Arlinghaus *et al.* 2002), measuring both the recreational value to the individual and the market value to the economy, enables estimates of the full benefits and total economic value of the fishery associated with stocking to be accounted for (Hunt *et al.* 2017). The value of socio-economic objectives achieved via enhancement depends on the functional relationship between catch-related satisfaction and marginal increase in catch rates (Camp *et al.* 2013), as well as the strength of any inherent stakeholder preferences for or against stocking as a management action (Baer and Brinker 2010, Arlinghaus *et al.* 2014).

Most cost effective comparisons of fish stocking do not account for indirect benefits (*e.g.* employment associated with hatcheries and transfers of fish) and non-use values in their calculations. If the benefits of a stocking program were positive without these values, additional non-use benefits would only strengthen the feasibility of a program that has already covered its costs (Rutledge *et al.* 1990). If the net benefits are negative, though, the magnitude of non-use benefits that would be required to make the program economically attractive can be gauged. The magnitude and sign of these net benefits will provide information on the minimal amount of non-use values that would be required so as to justify the stocking program as economically efficient. Garrod and Willis (1999) note that this is a common approach used to evaluate whether programs that include some non-quantified benefits, are economically justifiable. If socio-economic objectives via increased catch rates and related angler utility can be demonstrated (Anderson 1993, Anderson and Lee 2013) or increased effort and greater regional market activity result from augmented fish populations (Hilborn 1998, Camp *et al.* 2013), then stocking fish can potentially translate into net economic benefits.

There are a range of other limitations in using simplistic economic comparison techniques. Stocking program operations can take several years for benefits to accrue as released fish reach harvest size and require significant capital investment. Therefore discounted cash flow methods are required to determine the true feasibility (McCay *et al.* 2003). Analysis of the economic feasibility of fish stocking also needs to take account of possible losses in yield from the cost of negative effects on natural populations and ecosystems, such as genetic impacts on the wider stock (Hilborn 2004). This problem is an example of the negative externalities that are rarely included in economic assessments of enhancement (Lorenzen 2005). Failed enhancements carry social and economic costs because returns on investments are not realized and in some cases, significant externalities are imposed through ecological damage (Arnason 2001).

In this section key metrics on the economic feasibility of fish stocking are reviewed. These include post-release survival, the contribution of released fish to the fishery, cost efficiency, cost-benefit analysis and economic feasibility.

3.4.2.1 Lack of socio-economic studies

Although it is widely believed that stocking fish can increase fish numbers, examples of rigorous empirical evaluation demonstrating socio-economic performance are limited. The absence, or inadequacy, of monitoring programs following many stocking events has not allowed for evaluating the success of stocking and restricts improvements in further stocking techniques (Agostinho *et al.*

2010, Fitzpatrick *et al.* 2023). Robust post-stocking monitoring offers the opportunity to quantify the value of stocking and indicate if further stockings are worthwhile or necessary.

The lack of evaluation has also commonly led to the perception that stock enhancement efforts have been unsuccessful due to their inability to demonstrate a quantitative socio-economic contribution to the fisheries (Blaxter 2000, Chan *et al.* 2003, Molony *et al.* 2003, Taylor *et al.* 2017). In the absence of quantitative information, decisions to undertake fish stocking were often driven by the ability to produce fish for release (Hilborn 1999, Naish *et al.* 2007, van Porten *et al.* 2011) and community perception that introducing more fish into a system will result in more fish available to a fishery (Claussen *et al.* 2022). Key papers on responsible fish stocking all lament the lack of empirical data that has been collected, and highlight the importance of stocking programs to quantitatively demonstrate cost-effectiveness (*e.g.* Cowx 1994, Welcomme and Bartley 1998, Leber *et al.* 2005, Lorenzen *et al.* 2010). Such assessment is critical for effective fisheries management because it permits the efficient allocation of management resources and enables comparisons between alternative management strategies.

Suitable empirical data for accurately assessing the effects of hatchery releases are often lacking. One of the major limitations has been the ability to discriminate between released fish and wild fish (Mohler 2003). Identifying released fish provides the basis for quantitative assessment of fish stocking success because it enables information to be collected on released fish survival rates and their contribution to fisheries (Bell *et al.* 2005). Before the development of modern marking techniques, tagging systems were not applicable to the small, early life history stages released by hatcheries (Blankenship and Leber 1995), so historic stocking programs released fish that were not tagged or marked in a manner permitting the qualitative assessment of their impacts (Blankenship and Leber 1995).

Additionally, research-scale stocking programs released too few fish to be detected in open systems when evaluating changes in fisheries-dependent or fisheries-independent catch per unit effort as a measure of success (Blaxter 2000, Scharf 2000, Chan *et al.* 2003). Many early enhancement programs were also targeted at commercially harvested species, making it unlikely that small-scale stocking programs could impact catches with low numbers of stocked fish (*e.g.* Danielssen and Gjøsæter 1994). Even in modern stocking programs, often only a small proportion of released individuals are physically marked due to financial or logistical constraints (Kitada *et al.* 2018). Therefore, earlier stocking projects were rarely able to collect sufficient data for comprehensive socio-economic evaluation. Such limitations have been partially overcome in recent times by more sophisticated marking techniques (*e.g.*, chemical or thermal marking of otoliths, genetic identification) that have allowed researchers to mark larval and juvenile fish prior to their release.

The historic inability to understand the fate of released fish has constrained capacity to identify the underlying reasons why stocking programs might be under-performing or not meeting management objectives (Molony *et al.* 2003). The data collected now greatly improves evaluation opportunities and enhances understanding of stocking program success. Although understanding their contributions to fishery landings can indicate that hatchery fish have survived and entered the fished stock, a better estimate of stocking program success is the amount of increase in either total catch (commercial) or angler satisfaction (recreational) afforded by stocking. These are much harder to evaluate, but essential to know. For example, high fishery contribution rates from stocked fish, coupled with evidence that the total catch has not increased from stocking, is an indication that hatchery fish are merely replacing wild individuals and that stocking is unlikely to be successful or economically viable.

3.4.2.2 Contribution of stocking to fisheries

The ultimate contribution of stocked fish to a fishery is one of the most important management considerations for a stocking program, but has rarely been appropriately evaluated (Claussen and Philipp 2022). Stock enhancement programs must consider whether stocking produces additive effects on fish abundance or whether hatchery-reared fish replace wild fish in the system (Svåsand and Kristiansen 1990, Hilborn 1999, Leber 2002, Hunt *et al.* 2010). The effectiveness of stock enhancement should be evaluated by the contribution to the net increase in harvest value, user experience or abundance of the target species (Kitada and Kishino 2006). Instead, quantifying the contribution of stocked fish to fisheries and populations is frequently used as a measure to assess the relative success of stock recovery and enhancement programs.

Section 3.3 demonstrated that responsible stocking can result in substantial post-release survival in stocked individuals. However, it is unlikely that released individuals will provide a net benefit to population growth or future harvests unless appropriate resources, such as food and space, are available and/or the natural population is not recruitment limited (Hilborn 1998, 2004, Caddy and Defeo 2003, Bell *et al.* 2005). Monitoring and evaluation that detects poor stocking contributions to fisheries for specific waterways is important for management. Failure of stocking indicates that continued releases of hatchery-reared fish into these systems are ill-advised. Stocking resources can be reduced or diverted to waterways where stocking has shown the potential to positively affect the fishery. Reallocation of production would allow larger numbers to be stocked into waterways where stocking has proven to be successful and may further augment the fishery in these systems.

Contribution to a fishery typically refers to the ratio of stocked fish to all fish (stocked plus wild) of that species and year class in a water body and is an important component of responsible stocking (Blankenship and Leber 1995, Lorenzen *et al.* 2010). The contributions of stocked fish to fisheries are heavily influenced by the species' life history and its pattern of vulnerability to fishing, reinforcing the importance of biological attributes in species being released (Munro and Bell 1997, Taylor *et al.* 2005, Garlock *et al.* 2016).

Quantitative assessment of contributions to fisheries has occurred for the stocking of a wide range of species across multiple environments (Table 14 & Appendix A). Despite the significant application of stock enhancement and many large-scale release programs, the contribution of enhancements to global fisheries production has remained small (Lorenzen 2008). The majority of research evaluating the contributions of stocked species has been undertaken in Japan, Australia and the USA. Stocking contribution has been shown to vary across waters stocked in similar manners (Willis and Stephen 1987, Li *et al.* 1996) and across years within the same system (Fielder 1999, Walters and Martell 2004, Arbuckle 2006, Taylor *et al.* 2021). For example, stock augmentation of Red sea bream *P. major* has occurred on a massive scale and the releases appear to help maintain the catch in Kanagawa Prefecture and enhance it in Kagoshima Prefecture (Ungson *et al.* 1993, Kitada 1999). The contribution of released fish in the catch varies between prefectures, but has ranged from 14% (Ungson *et al.* 1993) up to as high as 64-83% in the inner Kagoshima Bay in Kagoshima Prefecture (Kitada 1999). In comparison, Tsukamoto *et al.* (1989) reported contributions of only 0.6-4.0% when stocking fry in the Oita Prefecture.

In the current literature review, contribution levels were found to varied greatly (0-100%) between stocked species, fishery type, release year and environment type, with an overall mean of 33.3 ± 22.2% across the 34 species examined by the 48 projects reviewed. Posterior mean contributions from stocked fish were typically highest in freshwater impoundments (41.1 ± 24.6%). Similar results were reported for stocking in coastal bays (36.5 ± 21.1%) and river systems (26.5 ± 26.3%), whilst slightly lower effects were found in estuarine systems (29.7 ± 18.4) and open coastal waters (26.5 ± 26.3%). Stocking Murray cod into impoundments resulted in the highest mean contribution (79.7 ± 28.4%) of stocked fish to a fishery, suggesting that fisheries in these systems are almost entirely dependent on stocking rather than natural recruitment (Forbes *et al.* 2016). The extreme example of this is for diadromous species such as Australian bass and Barramundi, which cannot breed in impoundments (Allen *et al.* 2002) and are thus 100% dependent upon stocking to support impoundment population. Stocking of relatively sessile marine invertebrates, such as sea urchins and scallops, can also lead to high stocked animal contributions, particularly in ranching scenarios (Table 14). The lowest mean contributions from stocking (<10%) were observed for riverine Brown trout *S. trutta*, Mulloway *A. japonicus*, Turbot *S. maximus*, Masu salmon *O. masou* and Pacific threadfin *P. sexfilis*, although for all of the species only a single relevant study was able to be found in the literature.

Very few evaluations explored whether stocking had an effect on total fish abundance by looking for absolute changes to the population size in the fishery, to determine if there has been an additive effect (*e.g.* Brennan *et al.* 2008, Hunt *et al.* 2010). Although studies that separate additive versus replacement effects are rare in the literature, they must be undertaken to assess the efficacy of stocking programs (Leber 2002) and determine if the replacement of wild fish by stocked fish is occurring (Hilborn 1999, Leber 2002). For example, Hunt *et al.* (2010) assessed Golden perch *M. ambigua* stocking in the southern Murray-Darling Basin and demonstrated that hatchery fish not only represented a substantial percentage of the adult fish, but that the abundance of fish had increased since stocking commenced. This provided strong evidence that stocking added to the total Golden perch population rather than replacing wild fish, and increased their availability for recreational anglers.

Globally there are several anecdotal examples where stocking has had an additive effect on target species populations. One of the major species stocked in Japan is the Red sea bream *P. major*. Stocking this species in a narrow bay in Kanagawa Prefecture, resulted in stocked individuals accounting for about 50% of the total catch of Red sea bream (Imai *et al.* 1994, Imai 1996). Annual commercial catches were virtually the same before and after the stock-enhancement program, but recreational catch increased more than 20-fold (Imai 1994). This suggests that stocking has was likely to have had an additive effect on the fish stocks. Surveys indicated that the benefit to commercial fishermen arising from the stock enhancement was estimated to be $912,700 in the prefecture in 1992, while the benefit to commercial guides running recreational party boats for anglers was estimated to be $6.66 million (Imai 1994).

In the USA, a major and successful Red drum *S. ocellatus* stocking program has developed in Texas. The proportion of stocked fish in the population can be as high as 20%, and there is general agreement that additive enhancement has taken place (Matlock 1990, McEachron *et al.* 1995, 1998, Tringali *et al.* 2008). In Norway, the large-scale release of juvenile European lobsters *H. Gammarus* resulted in a tag-recovery rate of 8% within a decade post-release (Agnalt *et al.* 2004). At the peak of the recaptures, released lobsters constituted around 50% of landings and the catch per unit effort of lobsters became markedly higher than in surrounding non-stocked areas. These results indicate that enhancement likely had an additive effect, rather than replacing wild recruits.

Case Study 3: Southern NZ scallop fishery stock enhancement program

In response to drastic harvest declines in the New Zealand Southern Scallop fishery, a program was established to capture wild spat and use them to seed specific fishing grounds. Wild spawn spat were collected on spat catching bags located in natural accumulation points within a bay, and seeded directly on the sea floor when of sufficient size (Drummond 2004). The program was self-funded by industry participants. In 1989-90, 150 million spat collected and seeded, and 220 t (meat weight) of seeded scallops were harvested (Arbuckle and Drummond 2000). The stock enhancement was further advanced through incorporation of a rotational fishing program, whereby the fishing grounds were divided into a number of zones. Each year a set number of these zones could be commercially fished and, following fishing, reseeded with scallop spat. By 1994, seeded scallops were estimated to have contributed 88% of the 850 t (meat weight) high-quality harvest (Anon 1998). Other fishery sectors also benefitted through an increase in the daily recreational bag limit from 20 to 50 per person, and a stocking allowance of up to 10% allocated for recreational and cultural fishing. Direct seeding was less effective in the closer inshore waters used by these groups. However, collection and redistribution of larger spat that had fallen off the collection bags resulted in successful supplementation. As the enhancement program grew in size, legislative management changes were introduced, and an industry owned company (Challenger Scallop Enhancement Company Ltd.) was founded to source and seed the spat. The success of the program is evidenced by the increase in commercial harvest from approximately 200 t (meat weight) in 1986, up to 700+ t (meat weight) in 2001, despite a significant reduction in the number of operators in the fishery.

In Australia, studies on the effectiveness of stocking programs for several freshwater and estuarine species have demonstrated strong, but highly variable, contributions from stocked fish to wild populations and recreational angler catch (Table 14). Recent research suggests that stocking often makes significant contributions to the fisheries catch and can have additive effects on the population size in some scenarios (Hunt *et al.* 2010, Crook *et al.* 2016, Forbes *et al.* 2016, Thiem *et al.* 2017).

Evaluation of several major Murray-Darling Basin stocking programs to enhance recreational fisheries in Victoria, indicated highly variable outcomes, with stocked fish representing from 11% to >99% of stocks of particular species in the enhanced fisheries (Ingram *et al.* 2015). Hatchery-reared fingerlings have been found to make substantial, although spatially and temporally variable, contributions to Golden perch *M. ambigua* populations, comprising between 18-100% of captured fish (Crook *et al.* 2016). In NSW, stocking has had variable results, with released Murray cod *M. peelii* comprising 0%-94% of the age-specific population in impoundments and 7%-15% in rivers. Stocked Golden perch comprised 23%-98% in impoundments, and 9%-14% in rivers (Forbes *et al.* 2016).

Chemical marking techniques have been used to show that stocking contributes significantly to Golden perch *M. ambigua* populations in both lakes and impoundments (47–90 %) and rivers (18–100 %) in southern parts of the Murray-Darling Basin, particularly where natural recruitment is low (Crook *et al.* 2010, Hunt *et al.* 2010). In Billabong Creek, where there was limited natural recruitment, stocking of Golden perch *M. ambigua* fingerlings over successive years resulted in a stocked fish contribution 79-100% of the catch and a four-fold increase in catch per unit effort (Crook *et al.* 2016).

In the Murrumbidgee River, the contribution from stocking of Golden perch was lower (29-37%), but also resulted in a four-fold increase in catch. This indicates that stocking most likely had a significant additive effect on the Golden perch populations in these systems. This concept is supported by the results of Hunt *et al.* (2010) who found that in three lakes with low natural recruitment, the contribution of stocked fish to the lake population ranged between 47-90%, and total abundance increased post-stocking (Hunt *et al.* 2010).

The comparatively low proportion of stocked Murray cod *M. peelii* and Golden perch *M. ambigua* in sections of Murray and Murrumbidgee Rivers suggests that these populations are primarily self-supporting through natural recruitment, and survival of marked fish is low into these waterbodies (Forbes *et al.* 2016). In contrast, stocked Murray cod and Golden perch formed a larger proportion of the population within Burrinjuck and Copeton dams. The virtual absence of unmarked golden perch in Copeton Dam indicates stocking is almost completely supporting this impoundment fishery (Forbes *et al.* 2016). Similarly, few unmarked Murray cod were identified in Copeton Dam, suggesting that natural recruitment was limited. The dominance of stocked fish in the impoundment populations suggests that fisheries in these systems are almost entirely dependent on stocking rather than natural recruitment. The variable contributions of marked fish demonstrate that adaptive, location-specific fishery enhancement strategies are required to maintain fisheries.

Several studies on Barramundi stock enhancement in north Queensland have demonstrated small to moderate contributions to recreational and commercial fisheries from stocking for recreational purposes. A long-term Barramundi *L. calcarifer* stock enhancement study in the Johnstone River found that, after only moderate stocking activity, stocked fish contributed between about 10 and 15% of the commercial and recreational catch respectively (Russell and Rimmer 1997,1998,1999, 2000, Russell *et al.* 2002). These figures indicate likely economic viability for this type of stocking. In the Dry Tropics, Barramundi stocking in freshwater impoundments and weir pools was not only critical to establishing (*e.g.* Ross Dam) and maintaining (*e.g.* Burdekin Dam) significant recreational impoundment fisheries that otherwise would not exist, but also contributed 3% to the wild-capture marine and estuarine fishery through downstream fish loss during overtopping events at weirs and dams (Leahy *et al.* 2022).

Stocking of hatchery reared Mulloway *A. japonicus* juveniles into NSW estuaries of varying habitat, resulted in mixed contributions to the receiving populations and harvest. The released fish yielded recapture rates of up to 0.2% in fishery independent surveys, but complete failure occurred during three stocking events (Fielder *et al.* 1999, Taylor *et al.* 2009). At Smiths Lake, stocked fish successfully entered the commercial and recreational fishery 18 months post release, and led to a 30-fold increase in the commercial Mulloway catch. Even with this increase, stocking was deemed to not be financially viable (Taylor *et al*. 2009). Experimental releases in the Georges and Richmond Rivers in NSW revealed stocked fish contributed 7% of angler's catch (Taylor *et al.* 2021), suggesting releases into river systems are more likely to be successful.

Case Study 4: Restocking Black bream in the Blackwood River, Western Australia

Long-term monitoring and assessment of a restocking program for the estuarine Black bream *Acanthopagrus butcheri* in the Blackwood River estuary of southern Western Australia revealed that the releases had a significant, positive effect on recruitment and population abundance (Cottingham *et al.* 2020). Natural recruitment in this system is very episodic and the problems posed by a marked decline in a fish stock were particularly severe given the stock is confined to an estuary and cannot therefore be naturally replenished from outside that estuary (Cottingham *et al.* 2020). The population of Black bream became severely depleted through over exploitation and deterioration of the catchment environment (Hodgkin and Hesp 1998, Prior and Beckley 2007). Local broodstock was used to hatchery culture 150,000 juvenile black bream which were chemically marked prior to release (Potter *et al.* 2008, Gardner *et al.* 2013, Cottingham *et al.* 2015). The restocking resulted in an abundance of juveniles approximately three times greater than that produced when natural recruitment of the wild stock was exceptionally high. By five years old, many of these restocked fish were making an important contribution to the commercial fishery, contributing as much as 53–74% of the catch (Cottingham *et al.* 2015), and also supporting the well-developed recreational fishery (Prior and Beckley 2006, 2007). The growth and maturity schedules of restocked Black bream were only slightly less that of the wild stock (Potter *et al.* 2008, Gardner *et al.* 2013, Cottingham *et al.* 2015).

Given the sporadic and infrequent recruitment events for this species in the Blackwood River, periodic releases of cultured fish are likely to benefit the stocks and help sustain recreational and commercial fishing and the commercial fisher in the estuary. Overall, the use of restocking overcame the need for politically and socially less acceptable options for maintaining the stock, such as the imposition of stringent harvest controls and fishing closures (Cottingham *et al.* 2020). The success indicates that hatchery-reared releases could also be employed to enhance sound stocks of other estuarine resident species, especially in intermittently open systems.

Stocking has also been demonstrated to make a significant contribution to invertebrate fisheries. In a study on stocked Blacklip abalone *H. rubra* in NSW, Chick *et al.* (2013) found that the released stock comprised between 88–99%, 75–88%, and 42–58% of the total abalone population at 108, 280, and 777 days post-release, respectively. It was demonstrated that the release of abalone to locations supporting depleted wild populations could result in a significantly greater abundance of total abalone populations after more than two years. Significantly greater numbers of all abalone (released and wild) occurred among release locations through time compared with control locations where abalone had not been released. The release of abalone stock also had no significant effect on the numbers of wild abalone and growth rates were similar between stocked and wild abalone, progressing to a harvestable size in 2.5-4.5 years. These results support the concept that restocking is an ecologically viable option in the management of depleted abalone populations, but financial viability also needs to be considered.

When planning stocking programs, fisheries managers should consider factors that influence the success of stocking, such as the impacts of connectedness of the system and changes in natural recruitment. The results from stocking freshwater systems in Australia indicate that there appears to be a relationship between increasing connectedness of habitats with natural recruitment and a reduction in the contribution of stocked fish to populations (Hunt *et al.* 2010, Crook *et al.* 2016, Forbes *et al.* 2016, Thiem *et al.* 2017). In lakes and impoundments, the contribution of stocked fish appears closely linked to the connectedness to a nearby river, which likely serve as the source for the wild fish population. These results support the concept that stocking programs can substantially augment wild fish populations where natural recruitment is low.

The contribution of stocking can fluctuate with natural year-class strength (Isermann *et al.* 2002, Forbes *et al.*2016). When natural recruitment is high, the advantage of rearing and stocking fish appears to be diminished (Cowx and Gerdeaux 2004). For example, the comparatively low proportion of marked Murray cod and Golden perch in riverine study sites suggests that these populations are primarily self-supporting through natural recruitment, and survival of marked fish is low into these waterbodies (Forbes *et al.* 2016). Freshwater fish stocking appears to provide the most benefit through the provision of substantial recreational impoundment fisheries, where natural recruitment in not occurring (*e.g.* Russell *et al.* 2013, Forbes *et al.* 2016). The dominance of stocked fish in the impoundment populations suggests that fisheries in these systems are almost entirely dependent on stocking rather than natural recruitment (Forbes *et al.* 2016). In rivers with high levels of natural recruitment, stocking may be inefficient, of limited benefit and potentially harmful (Hunt *et al.* 2010, Crook *et al.* 2016, Forbes *et al.* 2016, Thiem *et al.* 2017). However, in some systems the conditions required for successful recruitment may not occur because of river regulation or drought conditions, and stocking may be more effective. Similar results have been reported in a Striped bass *M. saxatilis* stock enhancement project in Chesapeake Bay, USA, where the success of released fish was strongly influenced by the level of wild production, suggesting that it was only feasible to augment Striped bass stocks in years of poor to average natural recruitment (Secor and Houde 1998).

Released fish sometimes do not travel far from the point of stocking (*e.g.* Leber *et al.* 1998, Nakagawa *et al.* 2004, Sakai *et al.* 2004, Chick *et al.* 2013, Russell *et al.* 2013). Therefore, stocking can have quite significant local fishery benefits, but more limited benefits across the whole fishery (Ziemann 2004). Release of 450,000 Black rockfish *S. melanops* into Yamada Bay, Japan, between 1989-1997, resulted in a 38.3% contribution of released fish to the local commercial catch (Nakagawa *et al.* 2004). The released juveniles increased the fishery catch, with almost all (99.3%) remaining in the limited area of the bay. In impoundments, where the natural movement patterns are disrupted or constrained, a scattergun release strategy (multiple release points) has the potential to result in a more uniform distribution of fish around the waterbody, particularly in large impoundments. Russell *et al.* (2013) found that most stocked Barramundi *L. calcarifer* were recaptured in the same general locale as they were released. There was some dispersion evident, but Barramundi abundance remained highest around release sites. A similar trend was observed in Pacific threadfin *P. sexfilis* stocked in Hawaii. Hatchery-reared fish accounted for 71% of captures from release sites, but only 8.7% in the broader fishery (Leber *et al.* 1998, Friedlander and Ziemann 2003). Released fish were recaptured on average less than 11.5 km from their release site, despite not being geographically constrained (Ziemann 2004).

Stocked fish can make significant economic contributions to commercial fishery harvests. This has been most well-studied in Japan, where substantial numbers of hatchery-reared fish are released annually to support their commercial fisheries. More than 25 million Japanese flounder *P. olivaceus* juveniles have been released annually in Japan in recent years with a mean market return rate (number of returned fish at market / number released) is 5.99% for all of Japan (Yamashita and Arataki 2010). These released fish have provided a mean contribution rate to the local markets of 11.7% (0.1-57.4%) of the commercial catch, producing a total fishery catch of approximately 800 million tonnes in recent years (Kitada and Kishino 2006, Yamashita and Arataki 2010). In Miyako Bay, Japan, the contribution rate of the released Japanese flounder to the total landings was between 16.5% and 52.7%, with a mean value of 33.6% by number and 27.7% by weight (Iwamoto *et al.* 1998). Across the entire survey period, the total landed value of stocked fish was ¥52.29 million (1997), representing 22.7% of the total income value. Likewise, stocking hatchery-reared Japanese Spanish mackerel *S. niphonius* was found to substantially contribute (29.9 t, ¥19.52 million 2005) to the fishery production in the Seto Inland Sea, Japan (Yamazaki *et al.* 2007).

Table 14 Summary for different species of the contribution of stocking releases to fisheries (± SD) and reported changes in fishery output or catch. Refer to Appendix A for the source references used to create the summary values.

Species	Environment	Fishery	Contribution to fishery (mean %)	Change in fishery
Abalone spp. *Haliotis spp.*	Coastal bay	Com	28.8 ± 12.5	n/a
Black rockfish *Sebastes schlegeli*	Coastal bay	Com	38.3 ± 24.3	Increased
Japanese flounder *Paralichthys olivaceus*	Coastal bay	Com	20.2 ± 9.1	1.2 x increase
Japanese scallop *Mizuhopecten yessoensis*	Coastal bay	Com	76.6 ± 20.4	n/a
Red sea bream *Pagrus major*	Coastal bay	Com	35.4 ± 35.5	Increased
Swimming crab *Portunus trituberculatus*	Coastal bay	Com	19.5 ± 5.0	n/a
European lobsters *Homarus gammarus*	Open coast	Com	55.3 ± 3.5	1.7 x increase
Japanese Spanish mackerel *Scomberomorus niphonius*	Open coast	Com	16.1 ± 19.4	Increased
Kuruma prawn *Penaeus japonicus*	Open coast	Com	13.4 ± 5.4	n/a
Masu salmon *Onchorynchus masou*	Open coast	Com	4.1 ± 1.1	n/a
Pink salmon *Oncorhynchus gorbuscha*	Open coast	Com	21.5 ± 35.5	n/a
Short-spined sea urchin	Open coast	Com	71.2 ± 12.7	n/a

Species	**Environment**	**Fishery**	**Contribution to fishery (mean %)**	**Change in fishery**
Strongylocentrotus intermedius				
Turbot *Psetta maxima*	Open coast	Com	3.7	n/a
Black bream *Acanthopagrus butcheri*	Estuary	Com, Rec	53.0 ± 14.8	n/a
Common snook *Centropomus undecimalis*	Estuary	Rec	41.2 ± 8.6%[1] 15.20 ± 6.2%[2]	2 x increase 1.1 x increase
Dusky flathead *Platycephalus fuscus*	Estuary	Com, Rec	37.5 ± 6.7	Inconclusive
Mulloway *Argyrosomus japonicus*	Estuary	Com, Rec	2.3 ± 3.6	Mixed - Nil to 30 x increase
Pacific threadfin *Polydactylus sexfilis*	Estuary	Rec	8.7	1.1 x increase
Red drum *Sciaenops ocellatus*	Estuary	Rec	20.0 ± 0.0	1.2 x increase
Sand whiting *Sillago ciliata*	Estuary	Com, Rec	48.0 ± 5.7	Inconclusive
Striped bass *Morone saxatilis*	Estuary	Rec	26.9 ± 11.7	n/a
Black crappie *Pomoxis nigromaculatus*	Impoundment	Rec	55.1 ± 21.0	n/a
Chinook salmon *Oncorhynchus tshawytscha*	Impoundment	Com, Rec	25	n/a
Golden perch *Macquaria ambigua*	Impoundment	Rec	49.4 ± 30.8	7.3 x increase
Largemouth bass *Micropterus salmoides*	Impoundment	Rec	20.6 ± 24.1	Mixed -Nil to 4.5 x increase
Murray cod *Maccullochella peelii*	Impoundment	Rec	79.7 ± 28.4	Supports fishery

Species	Environment	Fishery	Contribution to fishery (mean %)	Change in fishery
Rainbow trout *Oncorhynchus mykiss*	Impoundment	Rec	17	n/a
Barramundi *Lates calcarifer*	River	Com, Rec	12.5 ± 1.8	n/a
Brown trout *Salmo trutta*	River	Rec	1.5	n/a
Chinook salmon *Oncorhynchus tshawytscha*	River	Com, Rec	68	n/a
Golden perch *Macquaria ambigua*	River	Rec	40.2 ± 28.1	4 x increase
Murray cod *Maccullochella peelii*	River	Rec	34.1 ± 33.7	Inconclusive
Whitefish *Coregonus laveretus*	River	Rec	62.1 ± 5.0	n/a
Yellow perch *Perca flavascens*	River	Rec	26.0 ± 15.1	n/a

[1] High density stocking
[2] Low density stocking

3.4.2.3 Economic feasibility of stocking

The economic feasibility of fish stocking is a cost-benefit problem, where the benefits of increased catch, higher value or lower effort need to exceed costs of enhancement (Tlusty 2004). As has been highlighted in this Chapter, the mass release of hatchery-reared fish at considerable cost has not always led to increases in the abundances of target fishery stocks. Analysing economic feasibility and efficiency provides real-world information on fish stocking strategies to optimise their return on investment and enable comparison between alternative species and management strategies.

Different parameters often need to be considered when comparing the economic cost-effectiveness of fish stocking for recreational and commercial fisheries. The benefits realised in commercial fisheries are primarily market-based, and it is therefore relatively straightforward to calculate changes in producer surplus resulting from fish stocking (Whitmarsh and Pickering 2000). However, evaluating the marginal benefits in recreational fisheries requires greater consideration of both market (economic expenditure) and non-market (consumer surplus) impacts. Despite the differences in the way benefits are estimated, standard cost-efficiency analyses can be applied for the values generated.

Bioeconomic modelling is now regularly used to predict the likely economic success of a fish stocking program prior to full scale implementation. This form of economic feasibility analysis has formed the justification for the commencement or delay of several Australian projects and should form a core component of best practice use of fish stocking.

3.4.2.3.1 Willingness to pay (WTP) for stocking in recreational fisheries

There is strong belief amongst anglers that stocking fish will improve recreational fisheries (Hilborn 1999, Hasler *et al.* 2011, van Poorten *et al.* 2011). Angler willingness to pay (WTP) for stock enhancement has been used to evaluate the potential for establishing new stocking programs, expand existing stock enhancement projects, or refine stocking management strategies for recreational fisheries (Johnson *et al.* 1995). WTP values have been used within a framework of net present value to evaluate and prioritize potential enhancement projects (Dalton *et al.* 1998). The literature search identified twelve studies which directly related angler WTP to stock enhancement, and two studies which related WTP for improving angler catch for a range species that could be used to assess angler WTP for stock enhancement (Table 15). The majority of the WTP studies related to salmonid stocking in the northern hemisphere, but the results included examples from river, estuarine, marine and impoundment fisheries. It is difficult to directly compare recreational values from one study to another because the values are highly sensitive to site-specific variables such as existing catch rate, angler demographics or drivers behind participation (Duffield and Allen 1988, Krupnick 1993). Marginal WTP for additional fish is also inconsistent and heavily influences responses (Johnson and Adams 1989, Olsen *et al.* 1991, Waddington *et al.* 1994, Dalton *et al.* 1998).

Understanding user WTP helps estimate the likely socio-economic feasibility and public support for implementing stock enhancement programs for recreational fisheries improvement. There are numerous studies on the value of recreational fisheries, but few which focus specifically on the WTP or marginal benefits associated with fish stocking. Overseas, angler WTP for stocking programs has been reported for a number of species (Table 15), including Pacific threadfin *P. sexfilis* (Cantrell *et al.* 2004), Mulloway *A. japonicus* (Palmer and Snowball 2009), Black bass species (Waddington *et al.* 1994), Red drum *S. ocellatus* (Rhodes *et al.* 2018) and salmonid species (Carson *et al.*1989, Olsen *et al.* 1991, Johnson *et al.* 1995, Dalton *et al.* 1998). The results from these studies show there is typically high support from recreational anglers for fish stocking activities and consumer surplus typically exceeds the costs of undertaking fisheries enhancement activities. Marginal WTP values to improve fishing through stocking ranged from $2.47-$30.31 per day to $25.20-$150.13 p.a. (Table 15) Angler WTP to fish in stocked waterways ranged from $25.20 to $1,261 per annum.

For example, the maximum that fishers were willing to pay for stock enhancement of Mulloway *A. japonicus* in South Africa through an increase in the cost of a recreational fishing permit was estimated at $39 for frequent fishers and $25 for non-frequent fishers (Palmer and Snowball 2009). Adopting a license increase of $25 was projected to generate an additional $3 million from permit sales. The estimated costs to set-up and run a stock enhancement program ($1.5 million, van Rooyen *et al.* 2005) were significantly lower than this, suggesting that this strategy is likely to be an economically feasible management option (Palmer and Snowball 2009). Similar results have been reported for Pacific threadfin salmon *P. sexfilis* in Hawaii (Cantrell *et al.* 2004) and Red drum *S. ocellatus* in South Carolina (Rhodes *et al.* 2018).

In Queensland, two studies directly assessed angler WTP for fish stocking to enhance recreational fisheries. A combination of contingent valuation method (CVM) and travel cost method (TCM) estimated angler WTP to fish at stocked dams in Queensland to be $82-$1,262 per year, with anglers willing to pay an additional $26.50 to $60.00 per year for a 20% increase in catch rate in stocked dams (Rolfe and Prayaga 2007). These dams have no natural recruitment, so the value can be attributed entirely to fish stocking. Gregg and Rolfe (2013) found mean posterior angler WTP to fish in stocked impoundments was $212.12 per days fishing. Combining the TCM results with recreational fishing effort, resulted in estimated the total annual recreational fishing value for stocked impoundments in Queensland ranged from $0.19 to $20.26 million per impoundment, with a total of $109.71 million per annum for the 30 stocked impoundments in the SIPS program at the time (Appendix C).

Case Study 5: Lake Purrumbete salmonid stocking

The fish stocking program for non-native Brown trout *S. trutta*, Rainbow trout *O. mykiss* and Chinook salmon *O. tshawytscha* at Lake Purrumbete, south-western Victoria, Australia has created a put-grow-and-take recreational fishery. Hunt *et al.*'s (2017) evaluation of the enhancement program provides a valuable case study which informs our understanding of the cost-effectiveness creating new recreational fisheries through fish stocking. The average annual cost of stocking between 2007 and 2014 was estimated at $96,922 per year, including hatchery production and transport of fish to release. An angler creel survey estimated average observed angler expenditure of $81 per person per day. The observed economic expenditure (market value) associated with the stocking program was estimated to be $393,456 with a 4:1 BCR return on stocking investment. The additional willingness to pay (non-market recreational value) for the stocked fishery, was estimated to be an additional $411,250 ($94 per person per day) to $1,585,600 ($326 per person per day) with a 5:1 to 16:1 BCR return on stocking investment, depending upon the valuation rate used for the opportunity cost of travel time. Satisfaction and economic activity are key benefits derived from recreational fishing (Arlinghaus *et al.* 2002), so combining both the recreational value to the individual and the market value to the economy, gives a total net value of the fishery of between $0.76 and $1.88 million per annum.

Another indicator of angler WTP for fish stocking is the significant recreational fishing licence revenue paid by anglers around Australia each year. A large proportion of these license fees are directed towards stocking programs in both freshwater and marine systems (*e.g.* New South Wales, Queensland, South Australia, Victoria and Western Australia). For example, the Stocked Impoundment Permit Scheme (SIPS) in Queensland requires all recreational anglers fishing in designated stocked inland waterways to purchase a permit. The revenue raised is solely used to support the stocked fisheries for recreational angling purposes, and in 2020-21 the scheme sold 49,419 permits, raising $1.17 million for fish stocking (DAF 2022). Likewise, in Western Australia, the Marron ($50 p.a.) and Freshwater Angling ($50 p.a.) licences help support stocking in those fisheries (DPIRD 2022). In South Australia, anglers need to purchase a $33 p.a. permit to fish in the newly created stocked reservoir fisheries (SA Water 2022). Unlike non-market valuations such as CVM, the values from licence sales represent the minimum anglers are willing to pay for stocking and do not capture the consumer surplus.

Table 15 Willingness to pay (marginal benefit) for stock enhancement projects or additional fish in recreational fisheries

Study	Country	Fishery	Data collection year	Valuation method	Valuation context	Value (study units)	Value (AUD 2021)
Cantrell *et al.* 2004	Hawaii, USA	Pacific threadfin	1998	CVM	WTP for licence to fund stock enhancement	$13.70 USD p.a.	$38.98 p.a.
Cantrell *et al.* 2004	Hawaii, USA	Pacific threadfin	1998	CVM	WTP for stock enhancement to maintain current catch	$7.95 USD per trip	$22.62 per trip
Cantrell *et al.* 2004	Hawaii, USA	Pacific threadfin	1998	CVM	WTP extra fish	$10.05 USD per trip	$28.59 per trip.
Carson *et al* 1989	USA	King Salmon	1988	CVM	WTP extra fish	$28.10 USD p.a.	$81.11 p.a.
Gregg & Rolfe 2013	Australia	Stocked impoundments	2013	TCM	WTP to fish a stocked impoundment	$184.23 AUD	$212.12 per day
Harpman *et al.* 1993	USA	Rainbow trout	1991	CVM	WTP extra fish	$1.89 USD per trip	$4.86 per trip.
Hunt *et al.* 2017	Australia	Freshwater salmonids	2014	TCM	Opportunity cost to fish stocked impoundment	$84-291 per day	$93.96-235.51 per day
Hunt *et al.* 2017	Australia	Freshwater salmonids	2014	TCM	Economic expenditure to fish stocked impoundment	$72.23 per day	$80.80 per day
Johnson & Adams 1989	USA	Rainbow trout	1988	CVM	WTP extra fish	$6.65 USD p.a.	$19.20 p.a.
Johnson & Walsh 1987	USA	Salmonids	1986	CVW	WTP extra fish per day	$0.95 USD per day	$4.07 per day

Study	Country	Fishery	Data collection year	Valuation method	Valuation context	Value (study units)	Value (AUD 2021)
Johnson & Walsh 1987	USA	Salmonids	1986	CVW	WTP to fish stocked lake	$16.93 USD per day	$85.46 p.a.
Olsen *et al.* 1991	USA	Rainbow trout or salmon	1990	CVM	WTP extra fish	$14.81 - $54.84 p.a. USD	$40.54-150.13 p.a.
Palmer & Snowball 2009	South Africa	Mulloway *Argyrosomus japonicus*	2006	CVM	WTP for licence increase to fund stock enhancement	155 Rand p.a. frequent 100 Rand p.a. occasional	$39.07 p.a. $25.20 p.a.
Paulrud 2006	Sweden	Coastal	1998	CVM	WTP extra fish per trip	5 SEK p.a.	$43.79 p.a.
Paulrud 2006	Sweden	Coastal	1998	CVM	WTP extra kg of fish per trip	11 SEK p.a.	$96.33 p.a.
Paulrud 2006	Sweden	Coastal	1998	CVM	WTP double catch number	32 SEK p.a.	$280.23 p.a.
Paulrud 2006	Sweden	Put and take	1998	CVM	WTP extra fish per trip	42 SEK p.a.	$367.38 p.a.
Paulrud 2006	Sweden	Put and take	1998	CVM	WTP extra kg of fish per trip	58 SEK p.a.	$507.91 p.a.
Paulrud 2006	Sweden	Put and take	1998	CVM	WTP double catch number	44 SEK p.a.	$385.31 p.a.
Paulrud 2006	Sweden	River	1998	CVM	WTP extra fish per trip	531 SEK p.a.	$4,650.05 p.a.
Paulrud 2006	Sweden	River	1998	CVM	WTP extra kg of fish per trip	172 SEK p.a.	$1,506.23 p.a.
Paulrud 2006	Sweden	River	1998	CVM	WTP double catch number	160 SEK p.a.	$1,401.14 p.a.

Study	Country	Fishery	Data collection year	Valuation method	Valuation context	Value (study units)	Value (AUD 2021)
Rhodes *et al.* 2018	South Carolina, USA	Estuary	2005	CVM	WTP to fund ongoing stock enhancement	$19.32 USD p.a.	$36.79 p.a.
Rolfe & Prayaga 2007	Queensland, Australia	Stocked impoundments	2006	CVM	WTP for 20% increase in catch at stocked impoundments	$19.02 to $43.03 AUD p.a.	$26.54-60.04 p.a.
Rolfe & Prayaga 2007	Queensland, Australia	Stocked impoundments	2006	CVM	WTP for annual permit to fish stocked impoundments	$59.65-$904.40 AUD p.a.	$83-1,261 p.a.
Rosenberger *et al.* 2004	USA	Impoundment	2003	CVM	WTP to pay for private fish stocking	$29 USD p.a.	$65.12 p.a.
SIPS program	Queensland, Australia	Stocked impoundments	2022	Licence fee	WTP to fish in designated stocked waterways	$58.43 AUD p.a.	$58.43 p.a.
Waddington *et al.* 1994	USA	Trout	1991	CVM	WTP extra fish	$3.62 USD per trip	$9.32 per trip
Waddington *et al.* 1994	USA	Bass	1991	CVM	WTP extra fish	$3.72 USD per trip	$9.57 per trip
Wheeler & Damania 2001	NZ	Coastal	2000	CVM	WTP extra fish per trip	$1.61 - $19.76 NZD per trip	$2.47-30.31 per trip
Wheeler & Damania 2001	NZ	Coastal	2000	CVM	WTP extra kg of fish per trip	$2.40 - $5.79 NZD per trip	$3.68-8.88 per trip

3.4.2.3.2 Cost-effectiveness of stocking

In the reviewed literature, cost-effectiveness and return on investment were evaluated and reported using a variety of methods. NPV and Economic efficiency analyses focussed upon profitability for a certain business unit, whereas benefit-cost analysis had a much broader scope (Moksness *et al.* 1998, Kitada 2018). As Moksness *et al.* (1998) describes, NPV analysis for a planned stock enhancement includes only the economic factors presented in the model and is directly dependent on the realization of the stock enhancement: that is investments, income, and costs. A cost-benefit analysis would include, in addition, factors like infrastructural income/costs, new employment as a consequence of stock enhancement, multiplier effects on primary investments, environmental costs, better ground for new business, and so on. The cost-benefit approach also includes pricing of the environment or valuation of environmental goods. Therefore, the results from a cost-benefit analysis of a planned stock enhancement will show more positive figures than the NPV results do, mainly because of the expected high figures of the possible spin-off effects on other economic activities.

In many stocking programs, data on the environmental costs, annual costs for harvest, management and interest rates, and induced and indirect socio-economic benefits were not collected or unavailable, so NPV or Economic efficiency have been used as indices to compare the economic efficiencies between different projects and species. Measuring the market value of caught fish, cost to produce each stocked fish and number required to survive until capture in order to 'break even', was also commonly applied to evaluate the cost-effectiveness of fish stocking in commercial fisheries, without including the total benefits of the stocked fishery (Loomis and Fix 1999, Brown and Day 2002, Aphrahamian *et al.* 2003).

Case Study 6: Translocation of southern rock lobster to increase yield and market value

Variable growth of the Southern rock lobster *Jasus edwardsii* is the basis of ongoing spatial management challenges across the Australian fishery (McGarvey *et al.* 1999, Gardner *et al.* 2006). Lobster from deep water, offshore sites generally have slower growth, earlier maturation, and paler colouration than rock lobster from warmer, shallow water (Bradshaw 2004, Chandrapavan *et al.* 2009, McGarvey *et al.* 1999). This variation in demographic traits influences the yield and the economic drivers of the fishery. After the introduction of quota caps, fishers could no longer increase revenue by increasing their catch, so they looked to maximise the value of their catch by targeting larger and brighter red lobsters which have a higher value (Green *et al.* 2010).

A novel trial investigated how translocating slow-growing lobsters from deep-water regions to shallow-water, inshore areas has the potential to increase yield for the rock lobster industry in healthy fisheries in South Australia and Tasmania. Wild-sourced under-size lobsters were captured and moved to faster growth areas within their natural range. Growth of translocated individuals increased by 2-4 fold (Green *et al.* 2010) and they also developed a deeper red colour that resulted in a higher market price (Chandrapavan *et al.* 2009, 2011). Translocating wild stock within range avoids many of the problems encountered with other stock enhancement operations. Genetic and disease transfer risks do not exist because lobsters are moved within distances less than normal larval dispersal (Green and Gardner 2009). Translocated lobsters were found to remain at the release site, had equal survival to residents (Green and Gardner 2009) and increased growth rates (Chandrapavan *et al.* 2010). Egg production also increased, despite 30% of females delaying reproduction for the first year after translocation (Green *et al.* 2010).

The translocation trials developed into small-scale operations in Tasmania that are funded and managed by the commercial fishing industry, with operations in 2012 involving the translocation of 100,000 lobsters per annum (Green *et al.* 2013). The increased growth rates in the slow-growing, sub-legal portion of the population therefore increases the harvestable biomass of the stock. As a consequence, the total allowable catch was maintained 5% higher than would have been possible otherwise (Green *et al.* 2013). Several studies have estimated that translocating lobsters is economically feasible. Gardner and van Putten (2008) examined translocations by either charter vessels or by fishers retaining their sub-legal catch and releasing these on their return trip to port. Lower cost fisher translocations appeared feasible except for short distance translocations from deep to shallow water in the same region. Greatest net benefit occurred from long distance translocations between regions with extreme differences in growth (from SW to NW Tasmania). These operations required vessel charter and led to a net state benefit of $169,000 per 5-tonne trip, with IRR approaching 400%. The cost per kilogram gain in catch for translocation operations was estimated at less than $A3/kg, which was more than 5 times less than their alternative method to increase their catch (leasing quota from other operations at $16/kg). Bioeconomic modelling by Gardner (2012) suggests that translocation of 100,000 lobsters per annum would lead to increased catch rates and a reduction in harvest costs, with a NPV estimated at $47.4 million (7% discount rate, over 15 years, AUD 2012). This equated to a marginal increase of NPV of 22% over current management.

3.4.2.3.3 Economic efficiency of stocking

The contribution of hatchery fish in landings can be an index of the impact from stocking programs (Kitada 2020). Recapture rate, yield per released fish and economic efficiency have been used to report and compare on the economic performance of stock enhancement projects for commercial fisheries in Japan, Denmark, Indonesia and Norway (Table 16). Economic efficiency describes the ratio of net income to the release cost, excluding personnel expenses and expenditure for hatchery facilities (Kitada 2018). It is a quick and simple index that can be calculated at lower cost than full benefit-cost analyses because most of the data is either readily available or comparatively cheap to collect. However, economic efficiency does not report on the marginal benefit to the fishery, especially if stocked fish are leading to displacement of wild recruits.

The results from previous studies reporting economic efficiency (Table 16) generated a posterior mean economic efficiency ratio across all projects of 2.93 ± 6.77, with mean economic efficiencies for fish and invertebrates of 2.3 ± 20 and 5.17 ± 12, respectively. These encouraging results suggest that on the whole, releasing hatchery-reared fish and invertebrates is likely to deliver positive economic benefits for commercial fisheries. However, the results were highly variable, with large standard deviations of the means. Several strongly positive results had a disproportionate impact on the posterior mean values. If the median economic efficiency is considered instead on the mean, then the median across all projects reduces to only 1.6, and for fish and invertebrates 1.4 and 3.5, respectively.

The pooled economic efficiency results therefore need to be considered with caution, as these values are approaching the economic viability threshold. Once other costs are taken into consideration, many of the projects are therefore unlikely to be economically viable. From the 29 studies examined, six projects returned negative economic efficiency, indicating they were not profitable, a further 14 had economic efficiencies less than 2, and only seven had economic efficiencies greater than 5. The highest economic efficiencies were reported for stocking of Japanese scallops in Hokkaido (18 ± 5) and Chum salmon in Japan (11.7 ± 6.3), whilst the long-term Atlantic cod stocking program in Norway returned a very low efficiency value of -24.1 ± 21.3. Economic efficiency could be a cost-effective way to identify highly successful stock enhancement programs without the expense associated with full cost-benefit analyses. This approach could also be an effective index to monitor responses in fishery performance due to changes in stocking strategies. Information from programs identified as successful (high economic efficiency) could also be used to refine or design other stocking programs to increase their likelihood of economic success.

Overall, releasing hatchery-reared invertebrates was more than twice as economically efficient than releasing fish. This trend was potentially driven by the likelihood of dispersal from a stocking site, with fish species far more mobile and likely to emigrate than the stocked invertebrate species. The same pattern was also evident in the economic efficiency data amongst different invertebrates. More mobile invertebrates such as lobsters and crabs, returned lower economic efficiencies, whilst sessile species such as abalone and scallops had higher economic efficiencies and were more likely to have economically viable stocking programs. Mixed results were observed for prawns, with the costs for stocking Kuruma prawns higher than the fishery harvest, but in Chinese white shrimp stocking had a relatively high (7.1 ± 2.1) economic efficiency. The dissimilarity between the prawn species may be due to differences in the life-history habitat uses between the two species. Kuruma prawns tend to migrate offshore as they mature (Tanida *et al.* 2003), whilst Chinese white shrimp are more likely remain in nearshore and estuarine waters where they were released (Wang *et al.* 2020).

In an evaluation of Japanese enhancement projects, Kitada (2020) concluded that despite generally positive economic efficiencies, all cases of Japanese hatchery releases, except Japanese scallop, were probably economically unprofitable if the costs of personnel expenses, facility construction, monitoring, management and negative impacts on wild populations are taken into account.

Table 16 Economic efficiencies of stocking projects for commercial fisheries. Economic efficiency is calculated as the ratio of net income to the production and release costs. Adapted from Kitada (2018).

Species	Location	Recapture rate (%)	Yield per release (g)	Economic efficiency	Source
Atlantic cod *Gadus morhua*	Norway	2.0 ± 2.9	16.5 ± 24.2	-24.1 ± 21.3	Svåsand *et al.* 2000
Baltic cod *Gadus morhua callarias*	Denmark	n/a	n/a	6.0-9.0	Støttrup & Sparrevohn 2007
Barfin flounder *Verasper moseri*	Hokkaido, Japan	12.1 ± 0.6	181.5 ± 9.0	2.7 ± 0.1	Kitada 2020
Black rockfish *Sebastes schlegeli*	Iwate, Japan	11.8 ± 0.9	n/a	0.6	Nakagawa *et al.* 2004
Brown marbled grouper *Epinephalus fuscoguttatus*	Indonesia	10 cm 59-77 15 cm 64-82	n/a	3.6-4.8 1.2-2.0	Yulianto *et al.* 2019
Chum salmon *Oncorhynchus keta*	Hokkaido, Japan	4	120	9.6	Kitada 2018
Chum salmon *Oncorhynchus keta*	Hokkaido, Japan	3.6 ± 1.1	118.5 ± 9.0	18.9 ± 7.2	Kitada 2020
Chum salmon *Oncorhynchus keta*	Hokkaido, Japan	2.6 ± 0.8	85.5 ± 28.8	6.8 ± 2.3	Kitada 2019
Japanese flounder *Paralichthys olivaceus*	All of Japan	6.0 ± 6.5	n/a	n/a	Yamashita and Aritaki 2010
Japanese flounder *Paralichthys olivaceus*	Miyako Bay, Japan	12.7	n/a	1.2	Iwamoto *et al.* 1998
Japanese flounder *Paralichthys olivaceus*	Miyako Bay, Japan	14.5 ± 7.3	51.8 ± 24.2	1.6 ± 0.3	Okouchi *et al.* 1999 Kitada 2020
Japanese flounder *Paralichthys olivaceus*	Yamaguchi, Japan	n/a	n/a	9.1 ± 4.2	Hiyama & Kimura 2000
Japanese flounder *Paralichthys olivaceus*	Kagoshima, Japan	n/a	n/a	1.1 ± 0.1	Atsushi & Masuda 2004

Species	**Location**	**Recapture rate (%)**	**Yield per release (g)**	**Economic efficiency**	**Source**
Japanese flounder *Paralichthys olivaceus*	Kagoshima, Japan	2.4 ± 0.7	29.7 ± 2.6	1.1 ± 0.1	Kitada & Kishino 2006
Japanese flounder *Paralichthys olivaceus*	Iwate, Japan	13.5 ± 6.4	51.8 ± 24.2	1.6 ± 0.7	Kitada & Kishino 2006
Japanese flounder *Paralichthys olivaceus*	Fukushima, Japan	12.1 ± 4.8	n/a	0.9 ± 0.4	Tomiyama *et al.* 2008 Kitada 2020
Japanese Spanish mackerel *Scomberomorus niphonius*	Seto Inland Sea, Japan	15.0 ± 0.7	169.7 ± 8.3	1.0 ± 0.1	Obata *et al.* 2008
Japanese Spanish mackerel *Scomberomorus niphonius*	Seto Inland Sea, Japan	40 mm 4 100 mm 42	n/a	0.1-0.7 0.6-1.6	Yamazaki *et al.* 2008
Japanese Spanish mackerel *Scomberomorus niphonius*	Seto Inland Sea, Japan	15.0 ± 0.7	169.7 ± 8.3	1.0 ± 0.1	Kitada 2020
Masu salmon *Onchorynchus masou*	Hokkaido, Japan	fry 0.4 smolt 2.1	n/a	1.8 1.0	Miyakoshi *et al.* 2004
Red spotted grouper *Epinephelus morio*	Osaka, Japan	2.2 ± 1.0	7.7 ± 0.4	0.3 ± 0.0	Tsujimura 2007
Red sea bream *Pagrus major*	Kagoshima, Japan	8.0 ± 4.2	59.0 ± 27.2	5.0 ± 2.7	Kitada & Kishino 2006
Red sea bream *Pagrus major*	Sagami Bay, Japan	7.1 ± 2.9	54.9 ± 30.4	1.4 ± 0.3	Kitada & Kishino 2006
Abalone *Haliotis* spp.	Japan	12.2 ± 8.1	25.6 ± 19.1	3.5 ± 2.4	Hamasaki & Kitada 2008a
Chinese white shrimp *Penaeus chinensis*	Yellow Sea, China	7.2 ± 2.6	1.9 ± 0.6	7.1 ± 2.1	Wang *et al.* 2006 Hamasaki & Kitada 2008b
European lobster *Homarus gammarus*	Norway	6.2	37.2	0.5	Agnalt *et al.* 2004 Hamasaki & Kitada 2008b

Species	Location	Recapture rate (%)	Yield per release (g)	Economic efficiency	Source
Japanese scallop *Mizuhopecten yessoensis*	Hokkaido, Japan	34.5	69	4.6	Kitada 2018
Japanese scallop *Mizuhopecten yessoensis*	Okhotsk Sea, Japan	34.5 ± 10.2	60.9 ± 18	17.9 ± 5.3	Kitada 2019
Kuruma prawn *Penaeus japonicus*	Japan	2.8 ± 4.5	0.9 ± 1.5	0.7 ± 0.9	Hamasaki & Kitada 2006, 2008
Mud crab *Scylla paramamosain*	Kochi, Japan	0.9 ± 0.7	3.7 ± 3.0	1.9 ± 1.5	Obata *et al.* 2006 Hamasaki *et al.* 2011
Swimming crab *Portunus trituberculatus*	Kagoshima, Japan	1.2	1.5–33.6	n/a	Okamoto 2004 Hamasaki & Kitada 2008b

3.4.2.3.4 Cost-benefit analysis

The limited number of cost-benefit studies that have been undertaken have reported diverse results (Table 18), demonstrating fish stocking to be highly cost effective in some scenarios and economically unviable in others. Naish *et al.* (2008) suggested that stocking programs are generally not subjected to standard economic cost-benefit analyses because they offer multi-dimensional benefits to social, cultural, and political values, and thus are not strictly held to an expectation of financial profitability. Furthermore, it is difficult to quantify the possible negative impacts of hatchery production on wild populations, which would tend to erode economic benefits. However, the lack of cost-benefit analyses is likely an issue of political will rather than technical obstacles because this approach is commonly employed for other complex environmental policies (Anderson *et al.* 2020).

The literature search identified 30 studies which conducted some form of cost-benefit analysis. The results from these studies clearly demonstrate the heterogeneity of economic outcomes resulting from stocking to enhance both commercial and recreational fisheries. The posterior mean BCR for all studies was 12.46 ± 29.75. The large standard deviation is heavily influenced by a single outlier. Fish stocking conducted primarily to enhance commercial fisheries had a posterior mean BCR of 2.43 ± 3.28 indicating overall economic feasibility. However, half of the commercial fisheries stocking programs reported negative NPV or equivalent annual values (Table 18).

In the Norwegian Sea Ranching Program, the results might be positive from a biological point of view (contribution to fisheries), but it remains to be proven that the stocking is economically profitable (Moksness *et al.* 1998). An assessment of the economic performance found that NPV was negative for all species being released, across a broad range of recapture rates, juvenile production costs and market prices (Kitada 2018). These included Arctic char *S. Alpinus* , Atlantic cod *G. morhua*, Atlantic salmon *S. salar* and European lobster *H. gammarus*. For example, benefit-cost analysis of Atlantic cod cohorts released in Norway between 1990 and 1991, identified that all produced negative NPV estimates of between $-314,000 and $-718,000 per 100,000 fingerlings released (Svåsand *et al.* 2000). The two most critical conditions for profitable stock enhancement of Atlantic cod were the rate of recapture and the cost to produce juveniles. Sensitivity analysis suggested that the recapture rate of released fish would need to be over 32%, production costs for juveniles 16% of existing prices, or landed value nine times higher than typical market values. Therefore it is extremely unlikely that the Atlantic cod stock enhancement program will be economically viable. These results concurred with the findings of previous assessments by Sanberg and Oen (1993) and Moksness and Stole (1997). Some stock enhancement programs are bordering on economic viability. For Turbot *S. maximus* releases in North Zealand and Aalborg Bay, Denmark, the costs of producing and releasing hatchery-reared juveniles were slightly higher than the commercial value of their harvest most of the time (Støttrup and Sparrevohn 2007). An 8% recapture rate was required for their commercial value to match the costs for rearing and releasing the fish, and a few of the Turbot releases have reached this goal. This indicates the potential for economic viability of the stock enhancement program. Improvements in the post-release survival of released fish (increased recapture rate) or reducing the cost of hatchery-rearing will provide more consistent positive results and strengthen the economic viability of this stocking program.

There are some examples where releasing hatchery-reared juveniles to enhance commercial fisheries was clearly economically viable. Typically this occurred in species that had been well studied and the parameters required for successful stock enhancement were well understood and could be realised.

For example, Sproul and Tominaga (1992) determined that the long-term Japanese flounder *P. olivaceus* stock enhancement program in Ishikari Bay, Japan, was economically profitable. The economic returns from the harvest of hatchery-reared fish by local commercial fishermen, generated an estimated NPV (8% discount rate) of $5.4 million from the first 20 years of program operation, with a BCR of 3.15. Break even analysis indicated that if all independent conditions were held constant, the program would remain economically viable if landing prices could fall up to 78%, opportunity cost rise by up to 72%, or fry survival after 12 months could fall to 16%.

Ungson *et al.* (2003) demonstrated that stock enhancement of Red sea bream *P. major* in Kagoshima Bay, Japan, was also economically viable. The operation produced juveniles for both stock enhancement and aquaculture. Production costs were greater than the sales price realized for fingerlings (ROI = 80%), but the harvest of released fish made the operation economically profitable. The annual net income for stock enhancement was $6.5 million, whilst the combined hatchery and stock enhancement program returned a net income of $6.1 million. These provided ROI of 184% and 165%, respectively.
There have also been several feasibility studies conducted to assess the economic viability of implementing new stocking programs (*e.g.* Case Study: Brown Tiger Prawns). The feasibility of stock enhancement for the Baltic cod *Gadus morhua callaras* fishery in Denmark was evaluated using cost-benefit analysis (Støttrup and Sparrevohn 2007). This analysis was based on the costs of producing 474 million first-feeding cod larvae that should result in 17 million 2-year old recruits (Støttrup *et al.* 2005). The predicted harvest and spawning stock biomass were estimated to generate economic return rates of 6.0 in the case of the cod dispersing within the Baltic and 9.0 for the case of cod remaining stationary within the release area. The costs of release were not included and may reduce this value by up to 50%. These positive results will be further reduced if the survival of released fish is poorer than wild fish. However, the feasibility analysis indicated that the concept merited further examination.

The bio-economics of Chinese white shrimp *Penaeus chinensis* stock enhancement in China has been evaluated by calculating the ratio of costs to benefits. Release of shrimp into Hangzhou Bay was estimated to yield a BCR of 5.2 (Xu *et al.* 1997) and a return ratio of 1:7 to 1:10 has been estimated for the Haiyangdao and Qinghai fishing grounds (Wang *et al.* 2006). However, it is not clear what costs were included in these calculations and whether the cost:benefit ratio provides an accurate estimate of the real economic returns from stock enhancement.

Case study 7: Potential for stock enhancement of the Exmouth Gulf Brown tiger prawn fishery

Historically, the annual catches of Brown tiger prawns *Penaeus esculentus* in Exmouth Gulf, Western Australia, have experienced large fluctuations, primarily because of natural environmental effects on annual recruitment. The average catch since 1995 has also fallen by about 100 t compared to earlier years of the fishery (Loneragan *et al.* 2004). Large fluctuations in commercial catches can lead to poor economic performance in fleet and processing operations, and erratic supplies to markets. A continuous supply of product allows the market share to be maintained and maximises the efficiency and profitability of the fishery. A bioeconomic model was developed to assess the economic feasibility of using stock enhancement to increase and stabilise the Brown tiger prawn harvest in the Exmouth Gulf prawn fishery.

Loneragan *et al.* (2001) estimated that 24 million 1 g prawns would need to be released into the Gulf for the fishery to achieve a harvest of 100 t of hatchery-reared prawns. Model simulations showed that at that scale of release, a relatively stable median annual harvest of 113 t of released prawns would be achieved in addition to the highly variable wild catch (Loneragan *et al.* 2001). Stocking would lead to approximately 4.52 million more prawns being harvested at a recapture rate of 18.8%, which would be comparable with catches of stocked prawns in Japan (~22-28%), but higher than that reported for Chinese prawn stocking (7-10%, Loneragan *et al.* 2004).

Outside of the infrastructure costs for ponds and raceways, Loneragan *et al.* (2004) estimated annual production costs for prawns to be released were between $1.16-1.56 million, with a median of $1.4 million. The marginal revenue for the enhanced stock ranged from $615 000 to $4.40 million, with a median of $2.03 million. This produced a marginal profit between - $0.90 million and $2.30 million, with a median value of a $255,000 loss. Note that these estimates of "profit" provide an index of relative profits, not net profit because the capital costs of investment were not included in the calculations. There was a 48% chance of making a profit from enhancement each year. The initial outputs from the bio-economic model therefore indicated that the probability of enhancement would be economically viable in Exmouth Gulf was low.

Ye *et al.* (2005) further refined the bioeconomic model, leading to revised estimates of annual production costs of $1.76 million and marginal revenue of $2.35 million. This produced a marginal profit between -$1.40 million and $4.56 million, with a median of $0.32 million. On average, the refined model predicted enhancement to make a profit, but there was still a 34% chance of losing money.

The economic evaluations from fish stocking programs for recreational fisheries were all positive, except for the pilot stock enhancement study on Dusky flathead *P. fuscus* and Sand whiting *S. ciliata* in the Maroochy River, Queensland (see Case Study). Economic benefits were typically substantially larger than the costs associated with producing and releasing fish, with a mean BCR of 21.65 ± 40.59. However, the analysis by Rutledge *et al.* (1989) on Red drum *S. ocellatus* returned an extremely high BCR (52-260), almost an order of magnitude greater than other studies. This value considerably inflated the mean pooled BCR for recreational fisheries. A similar stocked recreational Red drum fishery in South Carolina was assessed to have a BCR of only 4.65 (Rhodes *et al.* 2018), suggesting that the Rutledge *et al.* (1989) figures may be too high. If the Rutledge *et al.* (1989) value is omitted from calculations, the mean posterior BCR for recreational fisheries reduces by almost half to 11.83 ± 14.80, and the mean BCR across all studies becomes 7.41 ± 11.12.

Case Study 8: The Maroochy estuary fish stocking program

One of the first large-scale pilot programs for marine stocking in Australia, examined the release of juvenile Dusky flathead *Platycephalus fuscus* and Sand whiting *Sillago ciliata* into the Maroochy River estuary, southeast Queensland (Butcher *et al.* 2000). Stocking of 100,000 Dusky flathead and 335,000 Sand whiting occurred over a period of three years. The contribution of released fish to both commercial and recreational catches was significant. Hatchery-reared Dusky flathead were found to comprise 28% of the commercial catch and 47% of the recreational catch. Similarly, hatchery-reared Sand whiting comprised 52% of the commercial catch and 44% of the recreational catch. These significant values suggest that released fish likely contributed to an increase in the total population. However, several sizable fish kills of wild stocks occurred during the program, which likely confounded the results by creating vacant niches for the released fish that would otherwise not normally have existed. Unfortunately, the program was also terminated before the effects of the stocking could be fully realised over a sufficient time frame. Economic analysis suggested the program was not economically viable, with an NPV of between -$1.03 million and -$1.43 million depending upon the costings model used. The return cost to the community per fish was $30-42, depending upon the model. This analysis made no allowances for the indirect benefits, nor the impacts of the fish kills, which may have led to a more positive economic outcome. Similarly, recreational value was assigned per fish, rather than on experiential value, which is likely to produce different results. Advances in hatchery-rearing technology has lowered the hatchery production costs for many species over the last two decades (Rowland 2013, Kitada 2018) and significant decreases in the cost of fingerlings is likely to lead to positive NPV.

In South Carolina, Red drum *S. ocellatus* are a declared gamefish, restricting the fishery to recreational anglers only. A pilot estuarine stocking program was established to examine if stocking 2.9 million 300 mm long juveniles annually could address the issues of declining catch rates and increasing demand, despite rising regulatory constraints (Smith *et al.* 2004). Rhodes *et al.* (2018) evaluated the economic feasibility of the stocking trial by comparing angler WTP (consumer surplus) against explicit stocking costs for a 10-year period. The NPV for the stocking program was positive ($35 million), with a BCR of 4.65, despite net costs of $11.0 million. The NPV was positive for all years, even if a worst case scenario WTP (USD $20.05 per angler per year) was used in the sensitivity analysis. This suggested that the program would have been economically efficient relative to having no program in place, and was economically feasible. Comparable results for Mulloway *A. japonicus* stock enhancement in Australia could be expected, since both species exhibit similar life cycles, especially utilisation of estuaries before migrating to open waters as they mature. Palmer and Snowball (2009) conducted a study to establish the economic viability of a proposed Mulloway *A. japonicus* stock enhancement program using the willingness-to-pay method, and found that under contingent valuation the benefits of stocking this sportfish far outweighed the cost.

Case Study 9: Queensland's stocked impoundment fisheries

Anglers during the 1950's and 1960's saw Queensland's impoundments as aquatic deserts (Hogan 2000). The highly modified environments and limited connectivity with riverine systems prevented many native fish species from establishing strong populations within impoundments. As hatchery production technologies and strategies developed, stocking of hatchery-reared fish was identified as a potential solution to create impoundment fisheries.

A Stocked Impoundment Permit Scheme (SIPS) was introduced in 2000 to provide some funding assistance to the stocking groups which maintain and enhance impoundment fisheries. The scheme initially covered 25 impoundments, but has spread to 63 waterways. Funds raised from the scheme are invested in fingerlings for stocking. In 2020-21 more than 49,419 SIPS permits were sold, comprising 19,044 annual permits and 30,374 weekly permits, and generating a combined permit revenue of $1,168,156 (Fisheries Queensland 2022).

Several studies have investigated the benefits and value generated from stocking Queensland impoundments. In the early days of stocking Tinaroo Dam, the BCR for Barramundi *L. calcarifer* was estimated at 31:1, with the fishery projected to have an annual value of $6,200,000 (Rutledge *et al.* 1990). Rolfe and Prayaga (2007) also estimated the value of recreational fishing at three stocked impoundments using TCM, identifying daily individual WTP of between $59.65 and $904.40, generating total annual consumer surpluses between $1.07 million and $4.54 million per dam. Contingent valuation identified that angler's annual willingness to pay for a 20% increase in catch rate ranged between $19.02 and $43.05 per person per year, providing total consumer surplus value estimates of $122,361 to $391,767 per dam.

Gregg and Rolfe (2013) also quantified the significant socio-economic benefits for regional communities from recreational fishing visits generated by stocking conducted under the SIPS program. Using TCM, it was estimated that on average anglers spent $184.23 per day fishing, providing a total economic value across 31 SIPS dams of $95.3 million per annum. The conservative values of individual impoundments ranged from $0.12-$10.42 million per year (Appendix C), but was likely higher based on the Rolfe and Prayaga (2007) results (revised values in Appendix C). The SIPS program has now expanded to cover 63 stocked waterbodies and the total economic value is expected to be significantly higher. The values from the SIPS program can be entirely attributed the fish stocking, because the fisheries are 100% culture-based with typically no natural recruitment. The ongoing release of fingerlings was necessary to create and sustain these put-grow-take fisheries.

The greatest economic benefits were achieved for recreational fisheries in enclosed waterbodies (*e.g.* freshwater impoundments) with limited natural recruitment and where emigration of stocked fish was restricted (*e.g.* Rutledge *et al.*1990, Hunt *et al.* 2017). Johnson and Walsh (1987) applied the contingent valuation method, to estimate the value of the stocked recreational fishery at Blue Mesa Reservoir, near Gunnison, Colorado. The gross annual economic value to anglers was estimated to be $23.38 million, comprising $11.37 million representing trip expenditures by anglers with impacts on state and local economic development. The remaining $12.01 million was the net value to anglers above what they spend. Costs for managing and implementing the stocking program were only $1.89

million per annum, demonstrating that the net value of the fishery at Blue Mesa was at least $10.12 million per year (net value to anglers less costs of stock and management) with a BCR of 6.34. The primary target species (Rainbow trout and Kokanee salmon) depend upon stocking to support their populations, with stocked fish representing <95% of fish in the population.

Buynak and Mitchell (1999) suggested that historic cost–benefit evaluations may be misleading in recreational fisheries because they do not always take into account the value of the stocked fish to the catch-and-release fishery. Monitoring was based upon harvest. This under-estimation has been overcome in contemporary studies through the use of non-market valuation techniques (*e.g.* TCM or CVM, Hunt *et al.* 2017).

Case Study 10: Stocking barramundi into the Johnstone River

Barramundi *Lates calcarifer* are a prized fish and there are extensive commercial and recreational fisheries for the species in northern Australia (Grey 1987), but there have been concerns over perceived ongoing declines in barramundi stocks (Russell *et al.* 2004). The Australian barramundi aquaculture industry is now well developed, and fingerlings can be produced cost-effectively (Rutledge and Rimmer 1991), making stocking a potentially viable management option in northern Australia. Barramundi were the first marine finfish to be stocked and monitored in Australia, beginning in 1986 for impoundments, and 1993 in open systems (Rimmer and Russell 1998). An experimental fish stocking programme centred on the Johnstone River in northern Queensland initially found that there found no significant difference in recapture rate between fingerlings released at 30-40 mm and 50-60 mm, with a mean recapture rate of 0.28% (Rimmer and Russell 1998). Subsequent research (Russell *et al.* 2004) investigated a further four size classes and found survival after release of 50mm, 70 mm, and 130 mm fish did not differ significantly, with recapture probabilities of between 0.09% and 0.17%. However, for fish released at 300 mm, the recapture rate increased considerably to 2.5%. Despite costing up to five times more to produce, the far higher recapture rate of the larger 300 mm fish, made stocking that size the most cost effective strategy (Table 17). Break-even analysis indicated that the stocking programme in the Johnstone River is potentially beneficial to the local community and the fishing industry, and depending on the size of stocked fish, only between 2 and 10% need to be caught to recoup the costs of purchasing fingerlings.

Table 17 Recapture rates and benefit-cost ratios for releasing different sizes of barramundi fingerling. Adapted from Russell *et al.* (2004).

Size class released (mm)	Production cost ($/fingerling)	Recapture probability (%)	BCR	BEA % recap req
50	0.62	0.09	1.43	2
70	0.80	0.14	1.78	2.5
130	1.37	0.17	1.22	4.4
300	3.06	2.56	8.36	9.8

Nine studies including cost-efficiency analysis were identified from Australian projects, of which four evaluated recreational fisheries and five commercial fisheries. The development of breeding technology for many of Australia's important recreational fisheries species has generated significant economic returns. For example, it was estimated that $40 million was invested in state-wide fish stocking in NSW, generating an economic contribution from stocking between $47 million and $133 million annually (NSW Fisheries 2003). In northern Australia, stocking of Barramundi *L. calcarifer* into impoundments has created several highly successful recreational fisheries which have been of significant economic benefit to rural communities. A cost-benefit analysis of the Barramundi stocking program in Tinaroo Falls Dam concluded that each dollar spent on fish stocking returned a potential $31 of economic benefit to the Queensland economy (Rutledge *et al.* 1990). Hamlyn and Beattie (1993) estimated that for every dollar spent on stocking in impoundments, $18 was spent in the local community by tourist anglers. With continued expansion in the popularity of impoundment fisheries, this ratio is likely to have increased. Boating, tackle and fishing media associated with impoundment fisheries have also expanded greatly in recent years with the development of these new angling opportunities.

Dredge *et al.* (2002) developed an economic model to evaluate a marine ranching operation of Saucer scallops *Amusium balloti* aimed at producing 100 tonnes of scallop meat per year through seeding suitable designated ranching areas. It was estimated that 178 million settled spat (wild collected) would be required to supply 35 million 10 mm juveniles for re-seeding. The model suggested that ranching would be highly profitable, with a 20-year project run (8% discount rate) generating IRR of 59%, BCR of 2.65, an NPV of $11.7 million, equivalent annual return of $1.186 million and a payback period of less than 3 years. The predicted IRRs and BCR were considerably higher than returns that have been modelled for other aquaculture operations and business ventures, and were linked to the high market price and rapid growth rate of Saucer scallops. The only significant social implications identified involved a potential perceived loss of social amenity associated with conservation and policing. There would, however, predicted to be considerable economic and social gains from successful marine ranching.
Not all Australian fisheries stocking studies have reported positive net economic benefits. Commercial re-seeding trials of Commercial scallop *Pecten fumatus* in Tasmania, whilst showing initial success, have been unable to produce reliable production even after significant financial commitment (O'Sullivan 2000). Likewise, small scale re-seeding trials at Jervis Bay in NSW were economically unsuccessful because of heavy predation (Heasman *et al.* 1998). Prince (2013) evaluated the economic feasibility of restocking Blacklip abalone *H. rubra* stocks in Western Australia depleted by disease, concluding that there is likely no advantage to restocking over natural recovery, unless recruitment in the depleted stock is subject to depensatory processes.

Stocking fish into artificially created waterways has the potential to create unique fisheries which can deliver significant socio-economic benefits to anglers and regional communities. In Australia, many impoundments are relatively isolated waterbodies, often with limited connectivity for fish to adjacent rivers and streams. These impoundments are typically created for flood mitigation, town water supply or irrigation, with little consideration given to fisheries value during construction and operation. Most of the large-bodied fish targeted by recreational anglers do not breed well in such lentic environments, resulting in very low fisheries quality within the impounded waters created by the dam. However, these waterbodies can have high productivity and are often well-suited to create put-grow-take recreational fisheries.

There are many examples of high value recreational freshwater fisheries in Australia that have been created and sustained through stocking in impoundments (*e.g.* Rutledge *et al.* 1990, Rolfe and Prayaga 2007, Gregg and Rolfe 2013, Hunt *et al.* 2017). In recent years, stocking has also helped establish highly successful new fisheries in previously unfished reservoirs in Adelaide and create a thriving Barramundi fishery in Lake Kununurra. More opportunities for new stocked impoundment fisheries are likely to arise from the construction of additional dams to supplement growing demand for water and the production of hydro-electric power.

Construction of dams can also create unique fisheries opportunities. Coldwater pollution from dam water releases typically has detrimental effects on recreational fisheries (Parisi *et al.* 2020). This should be mitigated wherever possible to minimise impacts on wild native fisheries downstream. However, where this is not possible, there may be potential to utilise the cold outflows to create stocked coldwater fisheries for salmonid species in areas where they typically can't persist due to water temperatures above their upper thermal tolerance threshold. Such a fishery was created in Lake Taneycomo on the Missouri River in the USA. The lake is fed by continuous hypolimnial discharge from Table Rock Lake for hydro-electric power generation and this has enabled establishment of a coldwater fishery in a warmwater fishery area (Fry and Hanson 1968). Stocked Rainbow trout *O. mykiss* thrive in the Lake Taneycomo, but their distribution is contained upstream by the Table Rock Lake dam wall, and downstream by thermal tolerance. The stocked trout have little impact on native fish, which cannot persist in the area due to the cold water temperatures. The put-grow-take fishery has become extremely popular, generating and estimated $27.6 million of net economic benefit to the local economy, or about 7% of all economic activity in the area (Weithman and Haas 1982). The BCR of the Rainbow trout stocking program at Lake Taneycomo was 22:1 for the local economy. Similar opportunities may arise from the construction of new dams in Australia.

There is also significant potential to expand the value of recreational impoundment fisheries through diversification of the stocked species available to anglers. A number of iconic diadromous or euryhaline fish species have yet to be evaluated for stocking in impoundments. Species such as Jungle perch *Khulia rupestris*, Big-eye trevally *Caranx sexfasciatus*, Giant trevally *Caranx ignobilis* and possibly even King threadfin *Polydactylus macrochir* are all endemic to coastal catchments in Australia and at times inhabit the freshwater reaches of coastal rivers and creeks (Allen *et al.* 2002). Development of stocked impoundment fisheries for such species has the potential to increase visitation and effort from both local and visiting anglers, value-adding to existing impoundment fisheries. Similar to currently stocked diadromous species, these fish require marine salinities for spawning, and thus their numbers in impoundments could be readily managed through appropriate stocking rates. It is recommended that research into the viability of stocking these species be undertaken.

Several issues regarding the economic feasibility for fish stocking need to be taken into consideration. In commercial fisheries, fishery access levels can play an important role in the economic viability of stock enhancement and marine ranching programs. This is a classic argument between the economical superiority of limited entry vs. open-access fishery policy. If stock enhancement delivers substantial economic benefits, increased numbers of fishers are likely to enter the fishery in enhanced areas unless there are restrictions in place. Assuming the increased number of fishermen remained, in each successive year fewer available net profits would be divided among more operators and diminished individual net profitability would result. It becomes clear to see that in an open-access fishery, stock enhancement projects are almost doomed to failure because their net profitability would eventually be eroded away by increasing fishing effort costs generated by new entrants attracted to the fishery. Limiting access to the enhanced fishery, especially if the enhancement costs are borne by existing operators, is the most economically successful strategy.

Additionally, whilst cost benefit analysis through BCR or NPV provides an indication of whether stocking resources are economically efficient, they do not provide an indication of projected net benefits that might have been generated by other fishery management programs using the resources available. This information is essential for fishery managers to help them decide upon the most appropriate enhancement strategies. For example, Prince (2013) quantified the comparative economic performance of different enhancement strategies using the impaired value of the individual transferable quota until stocks recovered, in an area where stocks were depleted by disease. The results provided insight into the potential economic outcomes of the different proposed management strategies in comparison to the base case of waiting for natural recovery.

Table 18 Economic results from past stock enhancement projects. All values have been converted to AUD 2021 to enable easier comparison. Appendix A contains the values in their original denominations. BCA – Benefit-Cost Analysis, NPV – Net Present Value, BEA – Break-Even Analysis, BCR – Benefit-Cost ratio.

Species	Location	Source	Analysis method	BCR	Equivalent annual value (AUD 2021)	NPV (AUD 2021)	Comments
Arctic char *Salvinus alpinus*	Norway	Moksness *et al.* 1998	NPV	n/a	n/a	-2.45 to -3.10 million	Small smolt (60-70g) Over 5 years Discount rate 10%
Arctic char *Salvinus alpinus*	Norway	Moksness *et al.* 1998	NPV	n/a	n/a	-2.85 to -4.15 million	Large smolt (300g) Over 3 years Discount rate 10%
Atlantic cod *Gadus morhua*	Norway	Moksness & Stole 1997	NPV	n/a	n/a	-0.70 million	Over 3 years Discount rate 15%
Atlantic salmon *Salmo salar*	Norway	Moksness *et al.* 1998	NPV	n/a	n/a	-1.62 to -1.99 million	Over 4 years Discount rate 10%
Barramundi *Lates calcarifer*	Tinaroo Dam, Australia	Rutledge *et al.* 1990	BCA	32 52	0.60 million 5.24 million	n/a	Single year 1st values – no multiplier 2nd value - 3.18 multiplier of travel cost to estimate indirect and induced benefit values
Barramundi *Lates calcarifer*	Australia	Hamelyn & Beattie 1993	NPV	18	n/a	n/a	Single year
Black sea bream *Acanthopagrus schlegeli*	Taiwan	Liao & Liao 2002	BCA	1.001	1,638	n/a	Single year <1% recap needed to cover stocking cost

Species	Location	Source	Analysis method	BCR	Equivalent annual value (AUD 2021)	NPV (AUD 2021)	Comments
Brown marbled grouper *Epinephalus fuscoguttatus*	Indonesia	Yulianto *et al.* 2019	BCA	3.55-4.82 1.19-1.99	2,931-4,335 1,837-3,086	n/a	Single stocking event of 1.2 yr Facilities costs not included
Brown trout *Salmo trutta* Rainbow trout *Oncorhynchus mykiss* Chinook salmon *Oncorhynchus tshawytscha*	Purrumbete Lake, Australia	Hunt *et al.* 2017	BCA	4.8-16	0.46-1.59 million	n/a	Values varied depending upon the opportunity cost of travel time used (0-100% wage)
Chinook salmon *Oncorhynchus tshawytscha*	USA	Wahle *et al.* 1974	BCA	4.2	2.53-17.89 million	n/a	Harvesting costs not included
Coho salmon *Oncorhynchus kisutch*	USA	Wahle *et al.* 1974	BCA	6.6-7.4	28.76-30.73 million	n/a	Harvesting costs not included
Dusky flathead *Platycephalus fuscus* Sand whiting *Sillago ciliata*	Maroochy River, Australia	Butcher *et al.* 2000	Outlay model and Total Cost model	0.07-0.09	n/a	-1.03 to -1.43 million	Over 5 years Compared models Influenced by fish kills
Freshwater salmonids *Oncorhynchus spp.*	Blue Mesa Lake, USA	Johnson & Walsh 1987	BCA	6.34	10.12 million	n/a	Single year Only annual comparison of rec value
Japanese flounder *Paralichthys olivaceus*	Hokkaido, Japan	Sproul & Tominaga 1992	BCA	3.15	n/a	5.43 million	20 yr program timeframe Discount rate 8% Facility construction costs not included

Species	Location	Source	Analysis method	BCR	Equivalent annual value (AUD 2021)	NPV (AUD 2021)	Comments
Largemouth bass *Micropterus salmoides*	Taylorsville Lake, USA	Buynak & Mitchell 1999	BCA	3.9	0.412 million (USD 1995)	n/a	Averaged over 5 ears
Mongolian redfin *Culter mongolicus*	China	Lin *et al.* 2021	BCA	2.6	n/a	n/a	Single year
Pink salmon *Oncorhynchus gorbuscha*	Canada	Boyce *et al.* 1993	BCA	n/a	-46.06 million	n/a	Over 30 years Undiscounted State benefits and costs only No recreational or subsistence values included
Rainbow trout *Oncorhynchus mykiss*	Missouri, USA	Weithman & Haas 1982	TCM Income multiplier	7.1 22	8.08 million 27.59 million	n/a	Compared valuation techniques
Red drum *Sciaenops ocellatus*	Texas, USA	Rutledge *et al. 1989*	BEA	52-260	513.02 million	n/a	BCR range depends on projected post-release survival (1% or 5%) 3 x multiplier for indirect & induced benefits
Red drum *Sciaenops ocellatus*	South Carolina, USA	Rhodes *et al.* 2018	BCA	4.65	4.23 million	35.02 million	Over 10 years Discount rate 3.5%
Red sea bream	Kagoshima Bay Japan	Ungson *et al.* 2003	BCA	1.65	6.12 million	n/a	Supplied fingerlings for stocking and aquaculture
Salmon spp.	British Columbia, Canada	Pearse 1994	BCA	0.61	n/a	-1,720 million	Life of program Discount rate 8% Facilities cost were irretrievable leading to negative result

Species	Location	Source	Analysis method	BCR	Equivalent annual value (AUD 2021)	NPV (AUD 2021)	Comments
Salmon spp.	British Columbia, Canada	Pearse 1994	BCA	1.6	n/a	479.21 million	Over 24 years 1993-2017) Discount rate 8% Facilities costs are foregone, so net benefits in continuing program
Sockeye salmon	Canada	Boyce *et al.* 1993	BCA	n/a	-$34.33 million	n/a	Over 30 years Undiscounted State benefits and costs only No recreational or subsistence values included
Striped bass	North Carolina, USA	Patrick *et al.* 2006	BCA	0.082-0.170	n/a	n/a	Over 15 years Not including non-use values
Brown tiger prawns *Penaeus esculentus*	Exmouth Gulf, Australia	Loneragan *et al.* 2004	BCA	0.96	-0.26 million	n/a	Single year Estimated positive return only 48% of time No capital costs
Brown tiger prawns *Penaeus esculentus*	Exmouth Gulf, Australia	Ye *et al.* 2005	BCA	1.33	0.32 million	n/a	Single year Estimated positive return only 65% of time No capital costs
Chinese white shrimp *Penaeus chinensis*	Zhejiang, China	Xu *et al.* 1997	BCA	5.2	n/a	n/a	Unclear what costs were included
Chinese white shrimp *Penaeus chinensis*	Xiamen Bay, China	Wang *et al.* 2006	BCA	7-10	n/a	n/a	Unclear what costs were included

Species	Location	Source	Analysis method	BCR	Equivalent annual value (AUD 2021)	NPV (AUD 2021)	Comments
Eastern king prawn *Melicertus plebejus*	Wallagoot Lake, Australia	Taylor *et al*. 2017	BCA	5.48	0.30 million	n/a	Used 2.3 multiplier for indirect and induced benefits
European lobster *Homarus gammarus*	Norway	Moksness *et al.* 1998	NPV	n/a	n/a	-19,916 to -28,451	Over 13 years Discount rate 10%
Greenlip abalone* *Haliotis laevigata*	Augusta, Australia	Hart and Strain 2016	BCA	1.7	3.55 million	54.13 million	Discount rate 6%
Saucer scallop *Amusium balloti*	Australia	Dredge *et al* 2002	BCA	2.65	1.19 million	11.74 million	Over 20 years Discount rate 8%
Short-spined sea urchin *Strongylocentrotus intermedius*	Hokkaido, Japan	Sakai *et al.* 2004	BCA	2.05-3.45	0.07-0.11 million	n/a	Facilities costs not included
All studies			**Mean ± SD**	**7.41 ± 11.12**			
Recreational fisheries			**Mean ± SD**	**11.83 ± 14.80**			
Commercial fisheries			**Mean ± SD**	**2.43 ± 3.28**			

3.4.2.3.5 Economic feasibility analysis and bio-economic modelling

The publication and use in decision making of realistic bio-economic models is an important step in putting enhancements into the fisheries management toolbox. Bio-economic modelling is key to appraising the potential of enhancement or restocking initiatives compared with other fisheries management measures and evaluating the cost–benefits of release programs (Lorenzen 2005, Ye *et al.* 2005, Gardner and van Putten 2008a,b, Prince 2013, Hart and Strain 2016). Bio-economic models provide a mechanism for integrating biological information with fisheries data and economic information to better assess the potential costs and benefits of release programs and predict the likely economic outcomes resulting from alternative stocking strategies (Hart *et al.* 2013, Taylor 2017, Tweedley *et al.* 2017). This approach is particularly effective for assessing the economic feasibility of new stocking projects and identifying optimal stocking strategies prior to commencement. Pilot studies are often required to define the biological parameters (*e.g.* post-release mortality, hatchery-reared growth rates etc.) where previous biological data on the target species or release location is unavailable.

Bio-economic modelling is a key element of undertaking responsible fish stocking and should be considered an essential part of best practice fish stocking. In Australia, this approach has been utilised to evaluate Southern rock lobster enhancement in Tasmania (Gardner and van Putten 2008a,b), stock enhancement of Brown tiger prawns in Exmouth Gulf (Loneragan *et al.* 2004, 2004, Ye *et al.* 2005), Greenlip abalone enhancement (Hart and Strain 2016), estuarine stocking in NSW (Cardno 2011), and estuarine enhancement of Eastern king prawns (Taylor 2017) and Western school prawns (Broadley *et al.* 2017).

For example, economic feasibility analysis was used in the NSW estuarine stocking program to assess the potential risks and benefits of the program (Cardno 2011). Key findings for both the qualitative and quantitative assessments were that, independent of location, all seven species assessed were likely to be economically feasible stock enhancement species. In particular, relative to the other species assessed the three species of crustaceans (Eastern king prawn *P. plebejus*, Giant mud crab *S. serrata* and Blue swimmer crab *P. pelagicus*) were seen as more likely to be viable. However, the motivations behind the capture of crustaceans may be significantly different from the four species of finfish. Non-finfish capture generally requires specialist equipment and is generally undertaken by a smaller proportion of the fisher population. Finfish stocking was likely to reach a broader fisher population, with Dusky flathead *P. fuscus* seen as the most viable species. The bio-economic model results supported the development of marine stocking guidelines and commencement of stocking activities.

Gardner and van Putten (2008 a,b) used a bio-economic model to examine Southern rock lobster *J. edwardsii* translocation scenarios in Tasmania. Two different methods of translocation were available, which affected both costs and benefits of the operation. The low cost system involved the capture of undersize lobsters from slow growth areas through normal fishing operations with these undersize lobsters then moved inshore. This approach had little marginal cost because fuel, labour and other costs were already sunk in harvesting operations. In the second approach, vessels could be chartered and funded to fish exclusively for the purpose of collecting animals for translocation. The bio-economic model found both approaches appeared economically feasible, but the higher cost chartered approach would provide a better economic return.

Bioeconomic modelling and feasibility analysis have also been used overseas to inform stock enhancement management decisions. In Taiwan, a cost-benefit analysis on the economic feasibility of stock enhancement for Black sea bream *Acanthopagrus schlegeli* showed that stocking is likely to be a cost-effective management technique for the recreational fishery (Liao and Liao 2002). Based on a realistic analysis, less than 1% recapture rate would be needed to cover the cost of a stocking program. The same result (< 1% recapture rate) was obtained by Russel and Rimmer (2001, 2002) using cost-benefit analysis on Barramundi *L. calcarifer* for the recovery of direct cost of stocking. In the case of lobster, using the simulation model LOBST.ECO, it has been shown that a recapture rate

Case Study 11: Bio-economics of Greenlip abalone enhancement in Australia

The potential of commercial-scale stock enhancement of Greenlip Abalone *Haliotis laevigata* was evaluated in Western Australia. Hart *et al.* (2016) found releases of juvenile abalone were able to stably increase abalone densities 400% above baseline, with the enhanced abalone cohort contributing 50% of the population. This demonstrated that the system was recruitment limited and could accommodate greater abalone biomass. No environmental effects from enhancement were detected other than the increase in abalone density, suggesting that as long as release densities are controlled within natural limits, successful stock enhancement can be attained for this species with minimal ecological impacts.

A bio-economic evaluation of commercial-scale stock enhancement was conducted to predict economic feasibility. Hart *et al.* (2016) reported that for the Augusta fishery, the optimal scenario involved an annual stocking rate equivalent to natural recruitment which was predicted to deliver:

1. 40% increase in spawning biomass (300t to 400t),
2. 85% increase in profitability ($1.15 million to $2.1 million),
3. 75% increase in annual gross value of product (GVP &2 million to $3.5 million),
4. 94% increase in NPV ($17 million to $32 million, 6% discount)

The bio-economic model was then extrapolated to the national level, and the optimal scenario involved an annual national stocking rate of 6.1 million 4 cm juveniles. This was predicted to result in:

- 25% increase in spawning biomass
- 120% increase in profitability ($12 million to $26 million),
- 60% increase in annual gross value of product (GVP $25 million to $40 million),
- 94% increase in NPV ($190 million to $420 million, 6% discount)

Following the success of the ranching project in Western Australia, ranching trials using the same approach were undertaken in Thorny Passage, South Australia. However, the Thorny Passage sites proved not to be commercially viable for Greenlip abalone ranching (Burnell *et al.* 2019). Each of the four trial sites exhibited low survival (5-31%) and poor growth rates compared to the Western Australian project. Stocked abalone suffered from high predation and low food availability, and the trial was terminated early. These results supported earlier assessments on Australian species which concluded there was limited potential for enhancement, citing carrying capacity issues, slow growth rates and high juvenile mortality as major limiting factors (Shepherd *et al.* 2000, Prince 2004, Chick 2010). Site-specific factors are therefore extremely important in determining the economic feasibility of stocking projects, and decisions need to be underpinned by comprehensive ecological understanding of the release sites.

above 23% is needed to make private sea ranching profitable (Borthen *et al.* 1999). For South African abalone *Haliotis midae*, recapture rates of greater than 10 to 15% are desirable for a modest profit (Cook and Sweijd 1999).

In South Africa, the economic feasibility of commencing stock enhancement of Mulloway *A. japonicus* was investigated using CVM to ascertain WTP, with an increase in the cost of a recreational fishing permit used as a potential vehicle of payment (Rhodes *et al.* 2018). The median value that fishers were willing to pay for a recreational fishing permit was R155 (South African Rand) for frequent fishers and R100 for non-frequent fishers. Analysis showed that a fee of more than R100 excluded up to 50% of anglers from the fishery, but that a fee of R100 excluded only 28% of recreational anglers and would generate an additional R12 million annually from the sale of recreational fishing permits. The estimated costs of set-up (R4 million) and running (R1-2 million p.a.) of a stock enhancement programme would be substantially lower than this, suggesting that stock enhancement may be an economically feasible management option. The feasibility analysis therefore supported further investigation into establishing the Mulloway stock enhancment progam.

3.4.3 Strategic considerations

To allow stocking to achieve its full potential as a fisheries management tool, an objective and strategic scientific approach is required (Molony *et al.* 2003). Best-practice stock enhancement should include scientific planning and evaluation of a stocking program against *a priori* objectives (Leber 2002), rather than undertaken simply to placate fishers or politicians (Blaxter 2000, Molony *et al.* 2003, van Poorten *et al.* 2011). It must be accepted that stocking will not solve all fishery problems, or be a suitable management approach in every situation (Cowx 1998). Stock enhancement will be of benefit only where the supply of juveniles regularly falls well short of the carrying capacity of the ecosystem and desired levels of recruitment (Bell 2006). The receiving ecosystem must have the capacity to support any additional fish, or replacement rather than population enhancement will occur. Stock enhancement must therefore be considered in conjunction with and parallel to other fishery management tools, and not in isolation, forming only one component of the overall fishery enhancement management strategy (Molony *et al.* 2003, Støttrup and Sparrevohn 2007).

Hatchery-reared releases for population enhancement are unlikely to be successful over the long term unless sufficient efforts are also made to regulate access and harvest rates (Kitada 2019). Successfully augmentation of fish populations in open-access fisheries may result in increased fishing effort and lead to greater fishing related mortality on wild fish (Baer and Brinker 2010). In commercial fisheries, diminished individual net profitability would result. In an open-access fishery, net profitability would eventually be eroded away by increasing fishing effort costs generated by new entrants attracted to the fishery (Sproul and Tominaga 1992). Similarly, in recreational fisheries the attraction of additional anglers and effort can result in only negligible marginal benefits for individual anglers. Therefore, stock enhancement will be most effective where restrictive regulatory controls are applied to manage both effort and harvest.

A comprehensive adaptive management framework for fish stocking is required to guide decisions about the most appropriate tools and approaches to use , and integrate with other enhancement measures to provide maximum benefits (Blankenship and Leber 1995, Bartley and Bell 2008, Lorenzen *et al.* 2010). Such a framework would help with decisions about how best to integrate associated priorities and rapidly encompass new scientific knowledge arising from monitoring and evaluation (Talley *et al.* 2018).

Bio-economic modelling is key to appraising the potential of enhancement or restocking initiatives relative to other fisheries management measures and evaluating the cost–benefits of release programs (Broadley *et al.* 2016). Such modelling enables consideration of whether economic returns are likely to exceed the costs of hatchery infrastructure, developing larval culture methods and producing hatchery-reared stock for release (Moksness *et al.* 1998, Ye *et al.* 2005, Taylor 2016). The publication and use in decision making of realistic bio-economic models is therefore an important step in putting enhancements into the fisheries management toolbox (Taylor 2016).

3.5 Summary

This Chapter has demonstrated that strategically planned fish stocking can make significant contributions to fishery catches and deliver substantial socio-economic benefits under the right circumstances. However, the mass release of hatchery-reared fish at considerable cost does not always lead to increases in the abundances of target fishery stocks, with few studies showing stocking to have an additive effect on total fish abundance. Inadequate monitoring and the inability to effectively discriminate between released and wild fish have restricted evaluations, as has a lack of scrutiny of the economic feasibility. In recent years there has been greater focus on research, monitoring and evaluation, which has provided clearer evidence that in enhancement operations, overall benefits can exceed costs. Conclusions regarding the potential of stock enhancement as a management tool can only be made if biological information is coupled with economic information to predict economic costs associated with stock enhancement relative to costs associated with alternative management approaches.
The majority of stocking projects have been conducted in degraded or overfished systems to recover fisheries, but there can also be opportunities for enhancement in healthy fisheries provided that the biology and population impacts of enhancement are well understood. Stocking can improve catch consistency, particularly where inter-annual natural recruitment is highly variable. In commercial fisheries, large fluctuations in catches can lead to poor economic performance in fleet and processing operations, and erratic supplies to markets. A continuous supply of product allows the market share to be maintained and maximises the efficiency and profitability of the fishery (Loneragan *et al.* 2004).

Stock enhancement through the release of hatchery-reared juveniles is not suitable for all species, therefore target species must be carefully selected and an appropriate long-term strategy developed to shift the fishery from one of catch only, to catch and enhancement. Species with unfavourable ecological profiles, high dispersal, low harvest value and high costs of hatchery production are less likely to be economically feasible to stock at a large scale. Focus should be on high-value species to achieve the best return on investment

The benefits of stocking vary with scale. Often significant benefits can be detected at the local scale (*e.g.* bay, impoundment or river reach), but limited changes are observed at the whole-of-stock scale. The mobility and dispersal of the released organism and the connectivity of the stocked environment both appear to play a significant role in the success and economic feasibility of stocking programs. The best results have generally been reported for resident species in closed systems, such as impoundments or bays, whilst few stocking studies in fully open systems or for highly mobile species have demonstrated economic viability. In commercial fisheries, releasing hatchery-reared invertebrates has been more than twice as economically efficient than releasing fish. High-value sedentary invertebrate fisheries have been recognised as good candidates for commercial scale stock enhancement, whilst popular angling species in impoundments or estuaries provide good candidates for recreational fishery stock enhancement. Stocking technologies and strategies need far more scientific development before stocking can be generally accepted as an economically effective fishery-management tool in coastal regions.

Angler willingness to pay has demonstrated the socio-economic feasibility and public support for implementing stock enhancement programs for recreational fisheries improvement, with consumer surplus typically exceeding the costs of undertaking fisheries enhancement activities. Anglers were willing to pay significant amounts for stocking activities to maintain existing fisheries, enhance fisheries and to create new angling opportunities (*e.g.* impoundment fisheries). The results suggest that expansion of the species and locations stocked in Australia is likely to be economically viable and have the potential to generate significant regional benefits.

The cost-effectiveness and return on investment from stocking have been evaluated using a variety of methods. In many stocking programs, data on the environmental costs, annual costs for harvest, management and interest rates, and induced and indirect socio-economic benefits were not collected or unavailable, so NPV or Economic Efficiency have been used as indices to compare the economic efficiencies between different projects and species. Overall, the NPV and Economic Efficiency analyses, comparing the direct economic benefits and direct production costs, produced economically marginal results, with only a few positive exceptions. If capital, staffing and management costs were included, most stocking projects would probably be economically unviable. Far fewer studies conducted comprehensive cost-benefit analyses encompassing a much broader scope, primarily because of the added difficulty and expense associated with determining the indirect and induced costs and benefits required to calculate the total economic value. Greater application of the cost-benefit analysis approach is expected to identify more stocking projects as economically viable as the broader economic benefits to the community are captured. This will enable better comparison between the relative impacts of stocking compared to other fishery enhancement measures.

Stocking practices in Australia can generally be considered to be undertaken responsibly, but require better evaluation, particularly of socio-economic factors. The conservative approach taken aligns well with the globally accepted "Responsible Approach" outlined by Blankenship and Leber (1995) and Lorenzen *et al.* (2010). The application of bio-economic modelling by Loneragan *et al.* (2004), Ye *et al.* (2005), Gardner (2012), Hart *et al.* (2013), Broadley *et al.* (2016) and Taylor (2017) in recent marine stocking programs demonstrated world best-practice. However, despite significant ongoing stocking in Australia's inland waterways, monitoring and evaluation of stocked freshwater species is still woefully insufficient and requires urgent attention. Developments in marking and tagging technologies (including genetic discrimination) now allow for better identification of hatchery-reared fish and should be applied to all stocked fish (Fitzpatrick *et al.* 2023).

Opportunities exist to improve the cost-effectiveness of stock enhancement programs. Cost efficiency gains can be achieved primarily through improved survival and fitness of released fish and reducing production costs. Research into these areas is likely to deliver positive economic returns on investment for stocking projects. If habitat in the area to be stocked is insufficient or degraded, it should be rehabilitated prior to the release of the hatchery-reared juveniles to improve survival, growth and carrying capacity. This is especially important for the nursery areas utilised immediately upon release. The optimal size of released juveniles and time of release are also critical considerations that affect survival of released fish and cost-effectiveness and should be determined prior to releasing juveniles. Acclimatization and pre-release conditioning programs to improve the survival, growth and reproductive success of hatchery-reared fish can be utilised to improve the rates of survival to harvest and/or spawning size, providing a better return on investment for stocking programs. For some species, collection of highly abundant wild larvae and post larvae can provide a source of juveniles for rearing programs that helps eliminate some of the genetic risks associated with captive spawning and greatly reduces the costs of collecting, holding and spawning broodstock. Alternatively, in other species there is potential to concurrently produce juveniles for stocking and aquaculture to reduce juvenile production cost. Implementing these approaches is likely to increase juvenile production costs and few studies have evaluated how these impact on economic feasibility. The key to developing all of these fields is the adequate and reliable marking of all stock being released to facilitate adaptive management to enable more cost-effective responses to fluctuating natural recruitment levels and environmental conditions.

The effectiveness of stock enhancement depends on how well stocking is integrated with other management strategies. Stock enhancement works best when combined with strong harvest regulations and environmental rehabilitation and protection. Construction of artificial habitats and restoration of lost or degraded areas can increase the carrying capacity and habitat quality for released cultured juveniles. If poor quality of insufficient habitat is present, releasing hatchery-reared fish is unlikely to deliver positive results and may waste management resources.

We still do not understand enough about the effects of stocking to use it effectively in some scenarios. However, the more quantitative approach now being taken in contemporary stock enhancement programs has highlighted the potential of this fisheries management technique when undertaken in a strategic manner. For stocking to become a practical fishery-management tool, cost-effective stocking strategies must be clearly defined. Stocking plans with protocols for critical stocking variables are needed to deliver safe and cost-effective outcomes.

Three key components should be incorporated to maximise the potential socio-economic feasibility of future fish stocking programs:

1. Pilot studies - Pilot releases should always be conducted prior to launching full-scale enhancement programs to inform development of optimal management strategies in a cost-effective way. Pilot releases identify enhancement capabilities and limitations and also provide the empirical data needed to plan enhancement objectives, test assumptions about survival and cost-effectiveness, and improve model predictions of enhancement potential. Thus, pilot-release studies that reveal ways to maximize survival of stocked fish without necessarily increasing rearing costs can improve cost efficiency in stocking programs

2. Bio-economic modelling - Bio-economic modelling should be considered essential for all new stocking proposals. Bio-economic models provide a mechanism for integrating biological information with fisheries data and economic information to better assess the potential costs and benefits of release programs and predict the likely economic outcomes resulting from alternative stocking strategies. Such modelling helps identify the main parameters influencing economic outcomes and can be used to examine their sensitivity to determine the necessary requirements for economic feasibility. Pilot studies are essential to inform the bio-economic models and enable development of optimal stocking strategies

3. Adaptive management framework - Active adaptive management is the single most important measure that can be taken to improve the potential for success in stocking programs (Leber

2004). Having an adaptive management framework enables the results from research, monitoring and evaluation to be rapidly incorporated into management plans and provides the capacity necessary for flexible responses to capitalise on new opportunities, address unforeseen threats and variations in natural recruitment.

Chapter 4. Habitat rehabilitation

4.1 General introduction

Wetland habitats play a vital role in supporting aquatic food webs and fisheries production (Weinstein and Litvin 2016). Wetlands can be defined as "*Areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water*[1]*.*"

Wetlands account for a wide variety of habitat types including rivers, shallow coastal waters and coral reefs. In Australia wetlands can be broadly categorised into three groups (Environment Australia 2001):

- Marine and coastal zone wetlands (*e.g.* bays, estuaries, intertidal zones).
- Inland wetlands (*e.g.* floodplains, riparian corridors, fens, lakesides).
- Human made wetlands (*e.g.* canals and drainage channels, stormwater retention basins).

The importance of wetland habitats can be both physical (habitat) and trophic, meaning that their value to fisheries production can greatly differ even among similar systems at different sites (Rozas and Zimmerman 2000, Minello *et al.* 2008, Minello *et al.* 2012).

Wetland ecosystems are highly productive and provide critical ecosystem services which support fisheries (Boyer and Polasky 2004, Brander *et al.* 2006, Mendelsohn and Olmstead 2009, Barbier *et al.* 2011). Vegetated ecosystems (*e.g.* freshwater marshes, seagrass meadows, salt marshes and mangrove forests) are widely cited as providing the nutritional input that underpin coastal fisheries production (*e.g.* Cole and Moksness 2016, Janes *et al.* 2020). For example, connectivity indices for mangroves, salt marshes and their associated channels explained up to 70% of the nearshore fisheries production on the Queensland coast (Meyneke *et al.* 2008). Wetland systems also act as essential spawning and nursery habitat, which makes them keystone features for fish production (Ronnbaack 1999, Beck *et al.* 2001, Bloomfield and Gillanders 2005, Sundblad *et al.* 2014, Taylor *et al.* 2016, Ley and Rolls 2018).

Many species exploited by fisheries rely on the various resources available in wetland systems for one or more life-history stages (Beck *et al.* 2001, Abrantes *et al.* 2015, Taylor *et al.* 2017), and this association may be either opportunistic or dependent (Elliott *et al.* 2007). Consequently, habitat loss and concomitant effects on these resources can adversely affect fisheries production, either through mortality and growth of early life-history stages, or through impacts on the productivity of exploited size-classes that rely on food webs supported by primary producers (Lenanton and Potter 1987, Taylor *et al.* 2018). Across many fish stocks, the types of habitats used by fish populations are therefore an important predictor of their long-term dynamics (Britten *et al.* 2016, Szuwalski and Thorson 2017). Maintaining and improving aquatic habitat function and quality is therefore critical to maintaining or enhancing fisheries productivity (Walker *et al.* 2004).

Globally, marine and freshwater wetland habitats are facing degradation and areal loss in many areas (*e.g.* Welcomme 2008, Waycott *et al.* 2009, Ye *et al.* 2013, Hamilton and Casey 2016, Huang *et al.* 2020). Drivers of these losses vary, but include eutrophication, development, land-use change, land

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1 *Ramsar Convention 1987*, Article 1.1.

reclamation, and climate change related warming (Hughes *et al.* 2017, Arias-Ortiz *et al.* 2018). Increasing severity and frequency of anthropogenic disturbances will likely increase loss and degradation of wetland ecosystems and reduce their ability to recover naturally (Perrow and Davy 2002, Stewart-Sinclair *et al.* 2021). Thus, habitat rehabilitation may be required to maintain the essential ecosystem services that support fisheries.

Habitat degradation has been suggested as the most important threat to the long-term recovery and harvest sustainability of exploitable fish stocks (Baird 1999). In Australia, this is often more so the case in freshwater and estuarine environments (Finlayson and Rea 1999). Habitat rehabilitation is therefore increasingly used around the globe to address the dramatic declines in the extent and function of many wetland ecosystems (Young 2000, Aronson and Alexander 2013, Perring *et al.* 2015). In Australia, the primary focus of fisheries management has been on controlling fishing pressure through harvest restrictions, with far less effort devoted to addressing stock production through rehabilitation of fish habitat. Often the option of habitat rehabilitation has been overlooked (Borg 2004). People want a quick fix to the perceived lack of, or reduction in harvestable fish available, and habitat rehabilitation can take time to return a positive fisheries response. However, there is a growing body of evidence that rehabilitating essential fish habitats can deliver cost-effective long-term enhancement of fish populations and fisheries production (Cappo *et al.* 1998, Rogers *et al.* 2019, Janes *et al.* 2020, Su *et al.* 2021).

Habitat rehabilitation describes activities aimed to repair or restore specific habitats and the ecosystem services they provide to ensure the survival of organisms, enhance the reproduction of specific populations, and strengthen overall ecosystem integrity. However, the terms “rehabilitation” or “restoration” have been used as broad phrases encompassing various levels of intervention in relation to habitats. In this Chapter, we use the following definitions:

Habitat rehabilitation - returning key ecosystem functions and services to a site without full restoration to a pristine condition (Hopkins *et al.* 1998). *i.e.* to enhance fisheries values of a site and assist natural succession to continue the recovery process.

Habitat restoration - returning a site to pre-existing conditions and implies a final objective to return all aspects of the previous system (Sheppard and McKinnon 1997). *i.e.* full restoration to a natural, pristine state.

Despite widespread degradation of wetland habitats, the foundations for recovery often remain available to rehabilitate or even restore ecosystems that have long lost their ecological function and productivity (Lotze *et al.* 2006). Although terrestrial rehabilitation has been practiced successfully for decades, wetland rehabilitation is still in development and is often short term, small scale, and expensive compared with terrestrial ecosystem rehabilitation (Blignaut *et al.* 2013, Costanza *et al.* 2014, Bayraktarov *et al.* 2020). Increasingly, ecosystem management strategies are focused on socio-ecological systems rather than ecological systems alone, enabling better understanding of the socio-cultural, economic, and institutional forces driving changes (Folke 2006). Rehabilitating key habitats can provide substantial long-term benefits for commercial and recreational fisheries, as well as a broad range of other ecosystem services (Rozas *et al.* 2005, Sheaves *et al.* 2014, Weinstein and Litvin 2016). However, large-scale habitat enhancement conducted from a fisheries perspective has experienced slow uptake for a variety of reasons. These include a lack of social and political confidence in the benefits of wetland rehabilitation, stemming from perceptions of low success rates, high implementation costs, low economic returns, and the risks of working in aquatic environments

(Bayraktarov *et al.* 2016, Saunders *et al.* 2020, Stewart-Sinclair *et al.* 2020). This perception of high risk and low returns is compounded by the limited duration of monitoring and evaluation for most projects, meaning that the full suite of benefits to fisheries are rarely detected.

“One of the most critical components of any enhancement effort is the ability to quantify success or failure. Without some form of assessment, one has no idea to what degree the enhancement was effective or, more critically, which approaches were totally successful, partially successful, or a downright failure”. Blankenship and Leber (1995).

It may be physically possible to restore a particular habitat, but there may be other reasons why it would not be done. Economic considerations are also essential to appreciate the full range of costs and benefits associated with habitat rehabilitation (Bayraktarov *et al.* 2016, Kittinger *et al.* 2016). This is further complicated for fisheries managers by the fact that, traditionally, habitat restoration has not been seen as a major role for the government departments that manage fisheries. Understanding the costs and benefits involved in wetland rehabilitation is therefore essential to its uptake and application as a fishery enhancement tool. Valuation of the potential outcomes that can be derived from habitat enhancement will assist with making an economic case in support of future work (Taylor 2016).

Accurate estimation of ecosystem service values allows their incorporation into decision-making frameworks, such as benefit cost analyses and economic impact assessments. Estimates of the value of different habitats can provide an objective basis for the prioritisation of enhancement and rehabilitation actions with other fishery enhancement options. However, the fisheries production that can be attributed to a particular unit of habitat (*e.g.* a wetland, a mangrove forest or a whole estuary) has proven difficult to estimate effectively (Abrantes *et al.* 2019). There can also be significant spatial variation in the values estimated. Janes *et al.* (2020) reported several fold differences between the capacity of coastal ecosystems to support fish production across Australian states. Care therefore needs to be taken if applying benefit transfer from one scenario to another (Woodward and Wui 2001, Kneib 2003).

To value the importance of wetland ecosystems for fish production, it is important to quantify and directly link ecosystems with fisheries data (Taylor *et al.* 2018). Estimating the economic benefit of habitat enhancement for fish and invertebrates requires determining the marginal value added to the harvest value of all exploitable species, which can be attributed to the enhanced habitat. It is also important to consider the contribution of all habitats at a landscape scale and across the entire life history of the species if the full importance of habitat enhancement to fisheries is to be understood or quantified (Janes *et al.* 2020).

Numerous studies have investigated the ecosystem service values for a wide range of wetland types, but the specific ecosystems services included have varied greatly. Valuations were rarely comprehensive, instead focusing on a few specific, easily quantifiable benefits. In this review, we have focussed specifically on studies which detailed quantified fishery-habitat links, so as to inform managers on the potential fishery outcomes that may be achieved from investing fisheries management funds in rehabilitation projects. Wetland ecosystems also provide considerable ecosystem services other than for fisheries (Barbier *et al.* 2011). Improvement of these services can value-add to rehabilitation efforts undertaken for fisheries purposes, but are often not the primary objective of fishery enhancement projects.

Wetland rehabilitation and restoration can be costly, ranging from \$139 ha^{-1} to \$17.3 million ha^{-1} (Table 19). Compared to terrestrial ecosystems, aquatic ecosystems, particularly coral reefs, seagrass meadows, and oyster reefs, are more expensive to restore (Mappin *et al.* 2021). The cost for restoration projects depends on the ecosystem type, the level of degradation, the economy of country where the restoration projects were carried out, and on the restoration techniques applied (Bayraktarov *et al.* 2016). For example, restoration of coral reefs and mangroves is significantly less expensive (up to 30 times) in developing countries compared to those with developed economies (Bayraktarov *et al.* 2016), and the total cost for coastal marine habitat restoration is 10–400 times higher than the maximum cost reported for the restoration of inland wetlands and freshwater systems (De Groot *et al.* 2013, Bayraktarov *et al.* 2016). Unfortunately, not all studies provide cost data in a consistent or comprehensive manner. The items included in stated costs often vary greatly, particularly for planning, administration and overhead expenses. It can therefore be difficult to identify the total costs for different rehabilitation techniques, making comparisons of the return on investment difficult.

Significant habitat rehabilitation efforts relevant to fisheries have been undertaken both globally and in Australia. Rehabilitation has primarily targeted coral and shellfish reefs, seagrass, mangroves, saltmarsh and river habitats. This has corresponded with a shift in emphasis towards ecosystem-based approaches to fisheries management (FAO 2003). This Chapter focuses on rehabilitation results for these habitats from a fishery enhancement perspective. Where possible, the focus was is on empirical socio-economic data from Australian studies to enable the outcomes from habitat rehabilitation to be compared against other fishery enhancement options. However, a more global context was necessary due to the limited quantitative socio-economic information available on historic habitat rehabilitation projects in Australia.

First, rehabilitation efforts in six key habitat types are described to provide a context of their extent of use. Second, ecological impacts and risks are discussed. Due to the broad extents of the field and the numerous reviews already undertaken, only a brief overview is provided. Third, empirical studies on the effects of habitat enhancement on fishery production and socio-economic values are summarized and compared between different habitat types. Finally, strategic use of habitat rehabilitation in fisheries management is discussed.

4.1.1 Mangroves

It is well recognized that mangroves support valuable fisheries driven by their high primary and secondary production and the nursery function of the complex intertidal habitat (Clough 1992, Kathiresan and Bingham 2001, Manson *et al.* 2005, Nagelkerken 2009). This is highlighted by findings that recruitment success and subsequent stock abundance may largely be determined by habitat availability (Bell and Nash 2004) rather than supply of recruits (Holbrook *et al.* 2000). The site fidelity of some species together with the essential nature of the ecosystem services provided to fisheries suggests that rehabilitation of lost or degraded mangrove areas may be a viable strategy for promoting stock recovery or enhancement (Le Vay *et al.* 2008).

Like many other wetland habitats, mangrove populations are on the decline, both globally and in Australia. This decline has been accompanied by reductions in productivity, biodiversity and the socio-economic value of artisanal and inshore fisheries (Costanza *et al.* 1997, Ronnback 1999). Attempts to value mangrove habitat–fishery linkages confirm that mangrove deforestation is contributing to fisheries decline and significant welfare losses (Barbier and Strand 1998, Sathirathai and Barbier 2001). Rapid mangrove loss, fragmentation and degradation create strong incentives for their restoration across their range to replace lost habitat and ecosystem services for fisheries.

Restoration of mangrove forests is possible, and has already been undertaken in many settings, but such efforts have typically been piecemeal, and many have failed to deliver the expected benefits (Field 1998, Kaly and Jones 1998, Alongi 2002). Despite this, a meta-analysis by Su *et al.* (2021) found that mangrove restoration can be cost-effective with positive benefit cost ratios under variable discount rates. The restoration age also effects the benefits to fisheries productivity. Mangrove rehabilitation provides higher ecosystem benefits over unvegetated tidal flats, while generally slightly lower than natural undisturbed mangroves (Ashton *et al.* 2003, Peralta-Milan and Salmo 2013). This may be due to biological differences or the lack of long-term studies which are needed to understand and capture the fully realised benefits. However, it has been found that in some instances that overall fisheries production was higher in restored mangroves than undisturbed mangroves, driven by higher prawn production, which more than offset the reported slightly lower fish production (Su *et al.* 2021). A study by Walton *et al.* (2007) also indicted that replanting of mangroves, even in mono-genus stands, was effective in restoring mud crab populations and catch rates, indicating recovery of an ecological function to a level equivalent to that of natural mangrove environments. This provides scope for the use of mangrove rehabilitation for fisheries enhancement. Restored mangrove ecosystems could be deliberately designed and engineered to maximise valuable ecosystem services for fisheries and be adaptable to climatic changes (Ellison *et al.* 2020).

In many cases restoration is largely achieved through the restoration of physical hydrological process. Restoring water flow and land elevation facilitates the natural transportation of mangrove propagules, and over time mangrove regeneration can occur with no planting. Planting, in contrast, can help to accelerate recovery rates, especially in places where natural recruitment may be low, but incurs a greater rehabilitation cost (Su *et al.* 2021).

Given the scale of mangrove degradation in Australia and their importance in fisheries production, there is significant scope for rehabilitation actions to have a substantial positive effect on fisheries. A model by Worthington and Spalding (2018) estimated mangroves are responsible for annually contributing more than 6.98 trillion finfish and 5 trillion invertebrates to commercial and recreational fisheries in Australia. The model also suggested that fisheries enhancement through rehabilitation of 336 km^2 of degraded or lost mangrove systems has the potential to contribute an additional 221 billion finfish and 1.4 trillion invertebrates annually. Such an increase would add significant value to Australia’s commercial and recreational fisheries.

4.1.2 Seagrass

Many commercially and recreationally important species have been linked to seagrass at some stage in their life cycle (Blandon and zu Ermgassen 2014, Gillanders 2006). The structural complexity and high primary and secondary productivity of seagrasses provides many ecosystem services, which support the spawning, survival and growth of fish and invertebrate species (Spurgeon 1999, Fonseca *et al.* 2002, Emmett Duffy 2006). Seagrasses provide essential nursery habitat (Connolly 1994, Loneragan *et al.* 2006) and support trophic linkages through food and prey production for recreationally and commercially important species (Dewsbury *et al.* 2016). Seagrass meadows also provides habitat and sometimes feeding grounds for species that inhabit coral reefs in their adult stages, or make diurnal treks between reefs and surrounding seagrass beds (Robblee and Zieman 1984).

The critical role seagrass plays in supporting important fisheries species generates significant value to the sector. For example, McArthur and Boland (2006) estimated that South Australian seagrass meadows had an annual economic contribution of ~$A114 million y^{-1} to fisheries. Clear relationships between seagrass beds and fishery species have been established (*e.g.* Loneragan *et al.* 2006, Blandon and zu Ermgassen 2014, Rogers *et al.* 2018, Janes *et al.* 2020) and changes in seagrass condition have been demonstrated to impact fishery production and have flow-on impacts on regional economies (Anderson 1989, Spurgeon 1999, Unsworth *et al.* 2010). Declining seagrass habitats are recognized as a significant threat to fisheries production (Jänes *et al.* 2019). With the continued loss of seagrass beds and potential implications of this habitat loss for fisheries resources, restoration of seagrass habitat is gaining importance and may offer a management tool for fisheries managers.

Enhancement of exploitable species within seagrass dominated ecosystems largely relies on the ability of organisms to naturally recruit via the water column (McSkimming *et al.* 2016). Several studies have found that restored and natural seagrass beds showed similar numbers and species of fish and shrimps within years of re-establishment (*e.g.* McLaughlin *et al.*1983, Fonseca *et al.* 1996). If seagrass meadows can be successfully restored, then harvestable species will find the new habitat, establish and survive long term. Evidence from Japan supports this notion. Kitada *et al.* (2020) reported that an increase in the wild Red sea bream (*P. major*) population size was caused by increased seagrass communities on the Red sea bream nursery grounds. The results suggest that rehabilitation of the degraded nursery habitat is indispensable to recover depleted populations.

Seagrass meadows are under increasing pressure from human development, with an estimated one third of seagrass meadows already lost globally since 1980 (Waycott *et al.* 2009). Seagrass losses in Australia follow the global pattern, with a reported loss of at least 291,783 ha, representing 5.5% of estimated areal extent, since the 1930s (Statton *et al.* 2018). These losses have occurred as a result of natural and human induced perturbations (Larkum *et al.* 1989, Kirkman 1997, Fraser *et al.* 2014, Thomson *et al.* 2015, Janes *et al.* 2020). Several large-scale declines have occurred in Shark Bay, Western Australia, Western Port, Victoria, and metropolitan Adelaide (Tan *et al.* 2020).

These losses can have major fisheries ramifications. For example, the estimated loss of 36% of seagrass meadows in Shark Bay resulted in declines of seagrass-associated fish populations, and closure of scallop and blue swimmer crab fisheries (Kendrick *et al.* 2019). Similarly, in Western Port Bay, Victoria, a decline of 70% of the seagrass cover was paralleled by a 40% decline in commercial fisheries harvest of species associated with that habitat (Jenkins *et al.* 1993).

Successful seagrass meadow rehabilitation has been rare and almost always at a scale orders-of-magnitude lower than the scale of loss. Australia has lost around 267,000 ha of seagrass, whilst successful seagrass restoration programs have been on the scale of <10's ha and 10's–100's ha of area revegetated (Stratton *et al.* 2018). However, if sufficient seagrass coverage can be re-established to meet the essential functional needs of a fisheries species, rehabilitation efforts have the potential to enhance local fisheries productivity.

The rehabilitation of seagrass meadows is expensive (Grabowski *et al.* 2012). Several restoration trials in Australia have utilized community volunteers to reduce rehabilitation costs, raise awareness and create a sense of ownership that encourages volunteers to return and donate more of their time (McKenzie *et al.* 2000, Tanner *et al.* 2014, McKenzie *et al.* 2017). Volunteer activities have included the collection of Posidonia shoots detached after storms for transplantation (https://www.operationposidonia.com/), community planting days on Kangaroo Island (Tanner *et al.*

2014), engaging recreational fishers in broadcasting seagrass seeds (Seeds for Snapper, WA, https://ozfish.org.au/seeds-for-snapper/), and collection of spathes for seed-based restoration in partnership with Indigenous Sea rangers in the Port of Gladstone (Central Queensland University 2020).

4.1.3 Shellfish reefs

Intact shellfish reef habitats are ecosystem engineers that help to structure the estuarine ecosystem, providing hard structure which serves as habitat for exploitable species of fish and crab and create nursery habitat rich in prey for juvenile fish and mobile crustaceans (Lenihan *et al.* 2001, Peterson *et al.* 2003, Grabowski *et al.* 2005). This high habitat value qualifies shellfish reefs as essential fish habitat (Coen *et al.* 1999, Coen and Grizzle 2007) and enables them to augment the production of commercially and recreationally valuable species in estuaries (Peterson *et al.* 2003, Grabowski *et al.* 2005, Crawford *et al.* 2019, McLeod *et al.* 2019). Shellfish reefs provide many ecosystem services besides fisheries production, including water filtration, food and habitat for animals, shoreline stabilization, and coastal defence (Nelson *et al.* 2004, Grabowski and Peterson 2007, Kellogg *et al.* 2013). They also increase the resilience and adaptive capacity of these communities to climate stressors through provision of thermal refugia for resident fauna (McAfee *et al.* 2020).

Shellfish reefs in Australia have followed the global pattern with wide-scale loss (Gillies *et al.* 2018). Historically the most abundant type of shellfish reef in Australia were constructed by oysters. However, in Australia natural oyster reefs are functionally extinct (meaning that they lack any significant ecosystem role), with more than 95% of such habitat lost (Beck *et al.* 2011, Diggles 2013). Fish produced on oyster reefs have significant value to coastal economies (Grabowski and Peterson 2007), and lost habitat caused by declines in oyster reefs have been linked to broader drops in fisheries productivity and economic value (Lipton 2004, Lotze *et al.* 2006, Airoldi *et al.* 2008).

Shellfish reef restoration has the potential reverse declining habitat and deliver many benefits and ecosystem services, especially for recreational and commercial fisheries (Coen *et al.* 2007). The primary goal of shellfish reef habitat restoration is the re-establishment of shellfish populations at self-sustaining levels similar to historic or natural shellfish communities (NSW DPI 2021). Restoration of shellfish reefs should not be viewed as a shellfish fishery issue (Frankenberg 1995). Instead, the aim of shellfish reef restoration is to support the growth and reproduction of target shellfish species, creating self-sustaining complex ecosystems that encourage the recruitment, survival and growth of fisheries species or their food and habitat resources (McLeod *et al.* 2020, NSW DPI 2021). In fact the greatest economic value of restored shellfish reefs is as habitat for exploitable fish and invertebrates, rather than for shellfish (Henderson and O'Neil 2003).

Realization of the extent of the loss of ecosystem services historically provided by shellfish reefs in Australia has led to recent efforts to restore them (Gillies *et al.* 2015, McLeod *et al.* 2019a, 2019b). Restoration of shellfish reefs has only gained management interest in Australia in the last decade. Coordinated efforts and large-scale restoration projects were kickstarted in Australia following a pivotal workshop in 2015 (Fitzsimons *et al.* 2015). Since then, Australian shellfish reef restoration projects targeting a variety of bivalve species and ranging in size from 0.1 ha up to 20 ha, are under way or have recently been completed. There are now more than 46 shellfish reef restorations underway (McAfee *et al.* 2022). Projects include Glenelg, SA, Kangaroo Island, SA. O'Sullivan Beach, SA, Windara Reef, SA, Gippsland Lakes, Vic, Port Phillip Bay, Vic, Oyster Harbour, WA, Pumicestone Passage, QLD, Noosa, QLD, Moreton Bay, Qld, Botany Bay, NSW, Port Stephens, NSW, Wagonga

Inlet, NSW, Derwent Estuary, TAS. Numerous additional projects have also been proposed, but are yet to commence.

Assessment of the success of oyster reef restoration showed that success varies across locations, depending on substrate used, overall project goals, and ecological conditions (zu Ermgassen *et al.* 2013). However, not all oyster reef restorations are successful for a variety of reasons including water quality, excessive sedimentation, toxic phytoplankton, and poor hydrodynamic flushing (Powers *et al.* 2009). If the original stressors that caused degradation have not been addressed, long-term reef re-establishment is unlikely to occur.

4.1.4 Saltmarsh

Salt marshes are one of the most productive ecosystems in the world (Mitsch and Gosselink 2008), and serve to maintain fisheries by boosting the production of economically and ecologically important fishery species (Zimmerman *et al.* 2000, MacKenzie and Dionne 2008, Taylor *et al.* 2018, Baker *et al.* 2020). They provide important nursery grounds for juvenile fish and invertebrates at the marsh edge (Raposa and Talley 2012, zu Ermgassen *et al.* 2021), with their complex and tightly packed plant structure supplying habitat that is mostly inaccessible to large fishes and birds, thus providing protection and shelter for the increased growth and survival of young fishes, shrimp, and shellfish (Boesch and Turner 1984). Saltmarsh habitats also include a diversity of producers that support considerable levels of primary and secondary production within the wider estuarine and coastal systems through the export of organic matter (Deegan *et al.* 2000, Becker and Taylor 2017, Raoult *et al.* 2018, Bennett *et al.* 2021). As elsewhere, Australian saltmarshes have been documented to produce organic materials (plant and animal matter) that are exported to coastal waters through tides, thus improving seascape fisheries productivity (Melville and Connolly 2003, Svensson *et al.* 2007).

The losses of saltmarsh around Australia can be credited to the historically prescribed low importance of saltmarsh leading to reclamation for industrial, agricultural, port and residential development (Laegdsgaard 2006). This valuable ecosystem is still under threat due to the expansion of urban areas, land reclamation, stock and vehicle access, and inputs of nutrients from the runoff of upland agricultural sites, freshwater influences, enriched groundwater, limited sediment supply, and sea level rise (Gedan *et al.* 2009; Deegan *et al.* 2012; Fagherazzi *et al.* 2019; Liu *et al.* 2021). These disturbances have already led to the loss of a considerable amount of salt marsh in Australia. Studies have also implicated the historic loss of saltmarsh habitats in the declining productivity of inshore fisheries (Barbier and Strand 1998, Creighton *et al.* 2015).

In response to the degradation and better recognition of the ecosystem services the habitat provides, substantial restoration efforts are being made to rehabilitate salt marsh in many areas of the world (Curado *et al.* 2014, Chen *et al.* 2017, Taylor *et al.* 2019, Xiao *et al.* 2020). Common restoration strategies use a combination of abiotic and biotic rehabilitation techniques (Billah *et al.* 2022). Abiotic techniques focus on re-establishing hydrology, tidal connectivity, and civil works to create suitable areas for saltmarsh plants, whilst biotic techniques involve planting to establish desirable vegetation communities. For example, in North Carolina, salt marshes are created through grading an upland site to intertidal elevations, followed by re-vegetation through planting with grasses and rushes (Craft *et al.* 2002, 2003). A major goal of coastal marsh restoration is to increase habitat for fishery species. Yet, coastal marsh restoration projects are seldom assessed for fishery support.

Restoration of saltmarsh in Australia is still a relatively new concept. As awareness about the ecosystem value of saltmarshes grows, so to do restoration efforts (Laegdsgaard 2006). In Australia, addressing the cause of disturbances and perturbations to allow natural regeneration has been a primary focus, so as to allow natural regeneration to occur. Actions such as removing barriers to tidal connectivity, land reshaping to restore tidal inundation, fencing to remove cattle from saltmarsh areas, diversion of stormwater away from saltmarsh, and weed removal are the most common rehabilitation methods for saltmarsh (Laegdsgaard 2006). Several Australian studies have examined the active transplantation of saltmarsh plants cultivated in greenhouses or taken from donor populations. The best results have been achieved where the environment has been pre-prepared for re-vegetation with saltmarsh plants (Burchett *et al.* 1998, Dick 1999). Knight (2018) documented 108 saltmarsh rehabilitation projects in Australia from Queensland and NSW. Unfortunately, rehabilitation efforts generally go undocumented and there is little measure of their success. Rehabilitation of the Hexam Swamp, near Newcastle, has been an exception to this, with detailed long-term research investigating both the ecological and fisheries outcomes from rehabilitation efforts (*e.g.* Boys and Williams 2012, Boys 2016, Hart *et al.* 2018, Taylor *et al.* 2018).

4.1.5 Coral reefs

Coral reefs support a wide range of ecologically and economically important species, by providing shelter space and substrate for smaller organisms, and food sources for larger epibenthic and pelagic organisms (Barbier *et al.* 2011). Healthy coral reefs support commercial and recreational fisheries as well as jobs and businesses through tourism and recreation. The value of coral reef fisheries can be significant for some economies (Cesar and van Beukering 2004, Zeller *et al.* 2007). For example, on the Great Barrier Reef in Queensland, the value of commercial fishing was estimated to be more than $104 million and recreational fishing more than $70 million (Deloitte Access Economics 2017).

Despite their great economic and recreational value, coral reefs are severely threatened by pollution, disease, and habitat destruction. Once coral reefs are damaged, they are less able to support the many creatures that inhabit them. When a coral reef supports fewer fish, plants, and animals, it also loses value as a tourist destination.

There is increasing concern about the progressive degradation of the world’s coral reefs (Bruno and Selig 2007). Coral reefs are currently facing global threats from climate change leading to higher ocean temperatures and acidification (Hoegh-Guldberg *et al.* 2007, De’ath *et al.* 2012). Local stressors include land-based pollution (Halpern *et al.* 2008), coastal development (Burke *et al.* 2011), outbreaks of the coral-eating crown-of-thorns starfish (Fabricius *et al.* 2010) and overfishing (Jackson *et al.* 2001).

While coral reefs in Australia have historically been well managed, recent declines in coral cover have triggered efforts to innovate and integrate intervention and restoration actions into management frameworks (McLeod *et al.* 2022). Despite facing serious threats, coral reef rehabilitation is surprisingly still in its infancy and very much focused on small scale, short-term technicalities (Hein *et al.* 2017). While these are necessary for improving rehabilitation processes, critical information as to whether coral rehabilitation can successfully increase reef resilience is currently lacking (Hein *et al.* 2019). Coral rehabilitation techniques can be categorised into three main methods: fixing the substrate, installing artificial reefs for natural and assisted coral recruitment, and transplantation of corals (Spurgeon and Lindhal 2009). Artificial coral reefs have also been suggested as a potential tool for improving the catch rate of a variety of fish and invertebrate species and creating habitats that enhance their recruitment and survival (Sherman *et al.* 2002, Perkol-Finkel and Benayahu 2004, Tran *et al.* 2019, Nguyen *et al.* 2022).

The cost of coral reef rehabilitation is currently extremely high, but the scale at which it can successfully be employed so far is very small. Additionally, rehabilitation will generally only be successful if the causes of reef degradation are known and have been reduced or removed (Edwards 2011, Omori 2011). This is a huge challenge for global impacts, and is also challenging even for local threats and stressors which can originate substantial distance away in the upper parts of coastal catchments.

4.1.6 Freshwater habitats

Rivers and other freshwater ecosystems support substantial fishery derived economic activity. Freshwater wetlands perform similar functions to those of tidal wetlands as providers of essential fish habitat. These wetlands provide food, shelter, nursery and spawning areas for many species targeted by recreational and commercial fishers (Jellyman *et al.* 2015). Freshwater aquatic plants, waterway banks, riparian vegetation and other microhabitats such as snags and riffle zones are all important for creating spawning sites, refuges and substrates for invertebrate prey (Jeppesen *et al.* 1997, Hickley *et al.* 2004). Riparian vegetation has a strong influence on water temperature and light filtration, in stream energy production and on the type and quantity of food matter and nutrients in the stream (Gregory *et al.* 1991). Riparian forests often play an important role in providing instream habitat of large wood, *i.e.*, snags (Lyons *et al.* 2000). Clearing this vegetation disturbs the natural ecosystems and can greatly affect fish populations (Price and Lovett 2002).

Habitat availability is not the only important factor influencing the value of inland fisheries. Migration is also central to the life history of many species (McDowall 1999). Numerous studies have demonstrated the negative effect that artificial barriers have on migratory fish and resident fish populations (*e.g.* Nislow *et al.* 2011, Gough *et al.* 2018, O'Hanley *et al.* 2020). Ecologically and economically important diadromous species spend part of their life cycle in freshwater and depend on connected habitats and appropriate migratory cues to successfully complete their life cycles (Skov *et al.* 2010). Other freshwater species are also migratory and require access to a range of habitats throughout their lifecycle. Weirs and barriers which prevent free migration can severely affect species abundance and even population viability (Harris *et al.* 2016). For example, Leahy *et al.* (2022) found that juvenile access to suitable freshwater habitats is important in sustaining the Barramundi (*L. calcarifer*) fishery and must be maintained. Barriers to juvenile fish movement into suitable freshwater nursery habits are therefore likely to be limiting fishery value. It may be possible to enhance the fishery's productivity by restoring fish passage to increase juvenile fish growth and survival.

Improving river connectivity through removal, repair, or modification of fish passage barriers has been demonstrated to deliver increased fish density (Gardner *et al.* 2013, Birnie-Gauvin *et al.* 2020), diversity (Catalano *et al.* 2007), and rapid colonization of formerly inaccessible stream reaches (Roni *et al.* 2008). Individual projects can restore access to kilometres of habitat, therefore properly performed fish passage restoration projects are one of the most cost-effective means to improve freshwater conditions for native fishes.

Rehabilitation of rivers and streams is an increasingly common approach to managing freshwater fisheries. River rehabilitation encompasses a wide range of specific management activities, from replanting riparian trees or fencing live-stock out of stream corridors to the removal of dams and full-scale redesign of river channels (Bernhardt *et al.* 2007). Remedial activities for fish focus mainly on reinstating lateral and longitudinal connectivity, recreating habitat diversity and channel morphology, improving flow regimes for fisheries purposes and improving water quality problems (Cowx and Welcomme 1998). Emphasis is placed on identifying bottlenecks to viable fish populations in the whole catchment, allowing the generation of habitat enhancement procedures that will aid rehabilitation of a natural balance. In lowland floodplain areas, reconnection of backwaters and ponds, or connection of adjacent gravel and borrow pits, which represent valuable spawning and nursery habitats, have been shown to increase species richness and relative abundance of fishes, especially if coupled with controlled flooding of the floodplain habitat (Jurajda *et al.* 2004).

In Australia, riparian land management is the most commonly undertaken river rehabilitation activity, followed by bank stabilisation, in-stream habitat improvements and channel reconfiguration (Bernhardt *et al.* 2005, Brooks and Lake 2007). Despite rapid increases in river restoration funding and activity throughout over the last 30 years, there has been limited evaluation of river restoration outcomes, globally and in Australia (Cottingham *et al.* 2005, Bernhardt *et al.* 2005). The need for assessing river restoration costs and benefits is nearly universally appreciated and is essential to broad-scale cost-effective uptake (*e.g.* Kondolf 1995, Kondolf and Micheli 1995, Bash and Ryan 2002, Downs and Kondolf 2002, Palmer *et al.* 2005, Ruiz-Jaen and Mitchell Aide 2005).

Rehabilitation of lakes largely focuses on shoreline habitat improvements, reducing nutrient input and overcoming the problems associated with eutrophication, including hypolimnic anoxia (Cowx and Gerdeaux 2004). Recruitment of fish populations can also be improved through reinstatement of shoreline habitat diversity and riparian and littoral zone vegetation (Jurajda *et al.* 2004). However, lake rehabilitation is less often undertaken in Australia compared to riverine project.

4.2 Impacts of habitat rehabilitation

4.2.1 Ecological impacts

In general, recovery of degraded wetland habitats is often challenging because of the dynamic nature of the system and is considered difficult to return to its historical pre-disturbed conditions (Billah *et all.* 2022). Habitat rehabilitation can deliver a variety of ecological benefits for fisheries and restoring environmental heterogeneity is one of the most important processes in maintaining and enhancing fish populations (Sass *et al.* 2017).

Aquatic habitats provide a wide range of ecosystem services, many of which underpin fisheries production. There are numerous mechanisms by which habitat change can affect the demography of fish populations (Vasconcelos *et al.* 2013). The population's carrying capacity is determined by the strength of intra- and interspecific density dependence and bottlenecks in habitat availability (Beechie *et al.* 1994, Milner *et al.* 2003, Brown *et al.* 2019). The available area of appropriate habitat or carrying capacity can therefore put an upper limit on the survival, abundance and growth of juveniles and adults (Sundblad *et al.* 2014). Habitat change may affect the population growth rate if it influences individual growth, survival of individuals at any life stage, or spawning production per individual (Brown *et al.* 2019). For example, loss of saltmarsh and wetland areas and barriers to fish passage can limit survival and growth of Barramundi (*Lates calcarifer*) in northern Australian river systems (Jardine *et al.* 2012, Leahy *et al.* 2022). Restoring nursery wetlands fish and passage will increase regional carrying capacity by enhancing access to these habitat types, improving juvenile survival and fishery productivity.

Ecological processes occur at varying rates depending upon the spatial scale and may govern the rate of ecosystem response and hence rehabilitation outcomes (Wiens 1989, Lake 2001). Different organisms or ecosystems respond at different rates relative to their growth and generation cycles. Trexler (1995) estimated that in the Kissimmee River, Florida, aquatic plants would recover in 3-8 years, invertebrates 10-12 years and fish 12-20 years. The response rate of ecological systems to rehabilitation activities is influenced by a number of factors, including spatial scale, level of degradation, rehabilitation activities and intensity, species and ecosystem composition, connectivity and abiotic environmental variables (Weins 1989, Arrington 1995, Trexler 1995, Pelley 2000, Lake 2001, Palmer 2010, Lyon *et al.* 2019). It is therefore important to consider ecological response timeframes when trying to capture the full impact of rehabilitation activities. Unfortunately, the typically long time frames of many ecological responses rarely align with the shorter windows of funding programs and our knowledge on the full ecological benefits that can be realised from rehabilitation are therefore somewhat incomplete. A recent survey of river rehabilitation in Queensland revealed that the ecological and fishery benefits from rehabilitation activities undertaken more than 15 years ago were still growing in scale and may not have even been fully developed or realised yet (Norris *et al.* 2020).

One of the issues facing this review was the relatively small scale of many past rehabilitation projects. Many have been at the research-scale and the area of habitat rehabilitated or restored is likely to be too small to have significant impacts on exploited species at the population scale. The exception to this is where rehabilitation projects have addressed a critical bottleneck (*e.g.* fish passage) that is specifically limiting population growth. However, studies have reported local benefits for target species, and if sufficient smaller rehabilitation projects are undertaken across the spatial extent of a population, then the aggregated habitat improvement may have implications at the population level.

The ecosystem services of rehabilitated sites do not always match that from undisturbed natural habitat. In a meta-analysis of 70 wetlands that had been rehabilitated 5–15 years earlier, it was found that compared to pristine reference sites, restored wetlands typically provided 16% lower supporting services and 22% lower regulating services than natural sites, even though they were able to match desired levels of provisioning and cultural services (Meli *et al.* 2014). Habitat rehabilitation for fisheries enhancement should ensure that rehabilitation efforts restore those ecosystem services that are necessary for increased fisheries production.

One key ecological question for habitat rehabilitation is whether productivity and the broader populations at rehabilitated sites increase or if only aggregation occurs. A large number of studies have been conducted at a wide range of spatial and temporal scales (Cowx and Welcomme 1998, Gregory *et al.* 2003, Nicol *et al.* 2004, Koehn and Nicol 2014), but there is still uncertainty about the scale of habitat rehabilitation required to generate broader population increases. Lyon *et al.* (2019) demonstrated that large-scale habitat rehabilitation can lead to fish population increases, not just aggregation. Four thousand four hundred and fifty large woody habitats were installed to rehabilitate instream fish habitat throughout 194 km of the Murray River between Lake Hume and Lake Mulwala and monitored for seven years. These efforts revealed a three-fold increase in the recreationally valuable Murray cod population in rehabilitated sites and no decreases at nearby control sites. This indicated production rather than aggregation was the likely outcome of the rehabilitation efforts and thus long-term benefits were predicted. Similarly, Debrot *et al.* (2022) demonstrated that the observed local fishing productivity increase observed following mangrove rehabilitation, was due to an increase in habitat productivity rather than attraction, which effectively would have meant long-term depletion of fish resources from adjacent areas.

Several studies have indicated that restored oyster reefs can enhance the overall carrying capacity of exploitable fish and augment of fish populations. Peterson *et al.* (2003) found that 10 m^2 of restored oyster reef habitat creates an additional 2.6 kg yr^{-1} of fish and large mobile crustacean production annually. This was thought to be because oyster reef habitat either enhances the recruitment rate of early life stages or enhances growth and survival by the provision of habitat with food resources and shelter from predators during some life stages. Similarly, a 1 ha trial oyster reef restoration project in Pumicestone Passage, Moreton Bay, Queensland, was found to significantly enhance the diversity and abundance of the fish assemblage (Gilby *et al.* 2020). The density of harvestable fish increasing by up to 16.4 times, whilst fish distributions across the broader landscape did not change, suggesting enhanced production rather than aggregation.

4.2.2 Socio-economic impacts

The economic feasibility of habitat rehabilitation for fisheries enhancement is a cost-benefit problem where benefits of increased catch, higher value or lower fishing costs, need to exceed the costs of rehabilitating habitats which improve fisheries productivity (Samonte *et al.* 2017). Cost-benefit analysis for habitat rehabilitation is rarely straightforward, especially in open systems, since benefits can be difficult and expensive to identify (Blaxter 2000). While acknowledging habitat rehabilitation provides a broad suite of ecosystem services, this Chapter focusses on benefits stemming from increased commercial and recreational fishing value. If these benefits are estimated to be greater than the costs of implementation, then a prospective rehabilitation project would be economically feasible from a fisheries perspective, and be suitable as a fisheries enhancement tool.

The benefits of habitat rehabilitation are by no means limited to the values presented here. Aside from benefits to fisheries productivity, restoration projects can also provide significant long-term benefits through the rehabilitation and strengthening of the ecosystem services restored areas provide (Samonte *et al.* 2017). For example, the rebuilding and restoration of oyster reefs can translate into the protection of critical habitat and infrastructure through wave attenuation, the improvement of habitat health through water filtration, and countless economic, recreational, social, and cultural benefits related to oyster harvesting. Despite the importance of these long-term ecosystem service benefits to coastal managers and stakeholders, not many studies have attempted their quantification. These limitations can affect the local, state, and commonwealth agencies' willingness to support further restoration work and the participation of stakeholders in habitat rehabilitation and enhancement initiatives.

The diverse array of benefits provided by ecosystems, the differing methods required to measure these benefits, and the bundled nature of the services produced by ecosystems, result in a complex tangle of economic means and ends (Costanza *et al.* 2008). Comparison of the rehabilitation costs for different habitats with the fishery benefits achievable from those habitats provides insight into the potential fishery-related economic benefits from rehabilitation activities. Several excellent global meta-analyses have been undertaken recently on the costs of rehabilitating different habitat types (see Table 19). Although rehabilitation costs are often highly site-specific, the median/mean values from these reviews provide reliable estimates for use in estimating the likely cost-benefits ratios for fisheries enhancement across various habitat types. The comprehensive (429 studies) global meta-analysis by Bayraktarov *et al.* (2016) found marine rehabilitation costs differed significantly between developed countries and developing countries, and thus the median results for developed countries have been presented in Table 19. All values presented here have been converted into 2021 AUD unless otherwise stated to facilitate easier comparison between project results. Similarly there have been several excellent reviews and meta-analyses of the fisheries values for different aquatic habitats (Table 19). These values provide indicative information on the benefits that might be achieved through habitat rehabilitation. It should be noted that only two studies were found which reported on the economic impacts of rehabilitation projects *i.e.* jobs, labour income and value-added output.

Table 19 Examples of economic value and rehabilitation costs for habitats important to fisheries. All values have been converted to AUD 2021 for ease of comparison.

Habitat	Location	Data type	Habitat value ($ ha^{-1} yr^{-1})	Rehabilitation cost ($ ha^{-1})	Source	Comments
Coral	Sri Lanka	Primary	226		Berg *et al.* 1998	
Coral	Indonesia	Primary	127		Cesar 1996	
Coral	Indonesia	Primary	7,394		Cesar 1996	
Coral	Jamaica	Primary	271		Gustavson 1998	
Coral	Australia	Primary		99,233	Kaly 1995	
Coral	Vietnam	Primary	5,665		Ngoc 2019	
Coral	Philippines	Primary	3,144		Samonte-Tan *et al.* 2007	Fisheries benefits only
Coral	Philippines	Primary	3,750		White *et al.* 2009	
Coral	Global	Review		2,271,203	Bayraktarov *et al.* 2016	
Coral	Global	Review		586,399	Bostrom-Einarsson *et al.* 2018	
Coral	Global	Review	542		Costanza *et al.* 1997	Mean value from global review
Coral	Global	Review		18,000-11,259,000	Grabowski *et al.* 2012	
Coral	Global	Review	626,671	26,668-17,333,455	Spurgeon 1999	
Freshwater marsh	Canada	Primary	7212.8		van Vuuren & Roy 1993	
Mangrove	Mexico	Primary	58,104		Aburto-Oropeza *et al.* 2008	

Habitat	Location	Data type	Habitat value ($ ha^{-1} yr^{-1})	Rehabilitation cost ($ ha^{-1})	Source	Comments
Mangrove	Thailand	Primary	188	452-1,581	Barbier 2000	
Mangrove	Thailand	Primary	571		Barbier 2003	Calculated from estimated loss of fisheries value due to mangrove clearing
Mangrove	Mexico	Primary	7,577		Barbier & Strand 1998	Commercial shrimp fishery value only
Mangrove	Malaysia	Primary	2,322		Chong 2007	
Mangrove	Indonesia	Primary	475		Christensen 1982	
Mangrove	India	Primary	9,598		Das 2017	
Mangrove	Sri Lanka	Primary	1,400		Gunawardena & Rowen 2005	
Mangrove	Fiji	Primary	5,867		Lal 1990	
Mangrove	Fiji	Primary	165-662		Lal 1990	
Mangrove	Vietnam	Primary		373-707	Ledoux *et al.* 2002	
Mangrove	Indonesia	Primary	3,510		Malik *et al.* 2015	
Mangrove	Australia	Primary		63,325	Melbourne Water 2013	
Mangrove	Australia	Primary	20,092		Morton 1990	
Mangrove	Kosrae	Primary	1,012		Naylor & Drew 1998	
Mangrove	Bangladesh	Primary	1,550		Rahman *et al.* 2018	
Mangrove	Bangladesh	Primary	1,338		Rahman *et al.* 2018	
Mangrove	Philippines	Primary	95		Samonte-Tan *et al.* 2007	Fisheries benefits only

Habitat	Location	Data type	Habitat value ($ ha^{-1} yr^{-1})	Rehabilitation cost ($ ha^{-1})	Source	Comments
Mangrove	Australia	Primary	994		Taylor *et al.* 2018	Commercial prawn fishery benefits only
Mangrove	Australia	Primary	60		Taylor *et al.* 2018	Commercial prawn fishery benefits only
Mangrove	Kenya	Primary	70		UNEP 2011	
Mangrove	Vietnam	Primary	166	139	Tri *et al.* 1998	Commercial fisheries value
Mangrove	Global	Review		48,469	Bayraktova et al 2016	
Mangrove	Global	Review		6,000-937,000	Grabowski *et al.* 2012	
Mangrove	Global	Review		664-637,687	Lewis 2001	
Mangrove	Global	Review	840	8,000-1,626,678	Spurgeon 1999	
Mangrove	Global	Review	1,233	1,616	Su *et al.* 2021	
Mangrove	Global	Review		5,431	Taillardat *et al.* 2020	Restoration costs - developed countries only
Mangrove	Global	Review		125,408	Taillardat *et al.* 2020	Restoration costs - all countries
Oyster reefs	Global	Review		83,083	Bayraktarov *et al.* 2016	
Oyster reefs	USA	Primary		302,576	Carlton *et al.* 2016	Recreational use and guiding only
Oyster reefs	USA	Primary		139,976	Calihan *et al.* 2016	3% disc. Fisheries benefits only. Commercial activities include oyster harvest
Oyster reefs	USA	Primary	13,077	7,955	Grabowski *et al.* 2011	3%, 25 yr
Oyster reefs	USA	Primary		148,040	Henderson & O'Neil 2003	

Habitat	Location	Data type	Habitat value ($\$$ ha^{-1} yr^{-1})	Rehabilitation cost ($\$$ ha^{-1})	Source	Comments
Oyster reefs	USA	Primary		100,106	Hicks *et al.* 2004	
Oyster reefs	USA	Primary	2,020	5,321,623	Kroeger 2012	Fish and crabs only, no oyster harvest
Oyster reefs	USA	Primary	14,871		Kroeger 2012	Fish, crabs and sustainable oyster harvest
Oyster reefs	USA	Primary	26,365		Lai *et al.* 2020	
Oyster reefs	Australia	Primary	27,533	223,933	Rogers *et al.* 2018	Fisheries and environmental benefits
Oyster reefs	USA	Review		439,889	Hernandez *et al.* 2018	
Oyster reefs	USA	Primary	7,160		Grabowski & Peterson 2007	
Oyster reefs	Global	Review	5,010	63,000-316,000	Grabowski *et al.* 2012	
Rivers	USA	Primary		50,498	Holmes *et al.* 2004	Value/cost per kilometre of river
Rivers	Switzerland	Primary	49,092,821	5,140,824	Logar *et al.* 2019	Value/cost per kilometre of river for all biodiversity and recreational benefits. Not fishery specific.
Rivers	USA	Review		64.395	Washington Trout 2004	Value/cost per kilometre of river
Salt marsh	Global	Review		83,465	Bayraktarov *et al.* 2016	
Salt marsh	USA	Primary	40,910		Bell 1997	
Saltmarsh	Australia	Primary	1,095		Janes *et al.* 2020	
Saltmarsh	Australia	Primary	2,500-25,000		Raoult *et al.* 2018	
Saltmarsh	USA	Primary		29,897	Rozas *et al.* 2005	

Habitat	Location	Data type	Habitat value ($ ha^{-1} yr^{-1})	Rehabilitation cost ($ ha^{-1})	Source	Comments
Saltmarsh	Australia	Primary	5,167		Taylor *et al.* 2018	GVP commercial fishery
Saltmarsh	Australia	Primary	484		Taylor *et al.* 2018	GVP commercial fishery
Saltmarsh	Australia	Primary	941		Taylor & Creighton 2018	GVP School prawn fishery
Saltmarsh	Australia	Primary	967		Taylor & Creighton 2018	
Saltmarsh	Global	Review		111,481	Taillardat *et al.* 2020	Restoration costs
Salt marsh	Global	Review		4,000-294,000	Grabowski *et al.* 2012	
Saltmarsh	Global	Review	533	5,333-426,670	Spurgeon 1999	
Saltmarsh & mangroves	Global	Review	1148.6		Costanza *et al.* 1997	Mean value from global review
Seagrass	Australia	Primary	35,619		Blandon & Zu Ermgassen 2014	
Seagrass	Australia	Primary	21,880		Janes *et al.* 2020a	
Seagrass	Australia	Primary	1,996		McArthur & Boland 2006	
Seagrass	USA	Primary	2,285		O'Higgins *et al.* 2010	
Seagrass	Europe	Primary		1,486,103	Perillo *et al.* 2009	
Seagrass	Australia	Primary	10,236	194,701	Rogers *et al.* 2019	Not fishery specific. Replanting cost
Seagrass	Australia	Primary	10,236	38,993	Rogers *et al.* 2019	Not fishery specific. Reseeding cost
Seagrass	Philippines	Primary	395		Samonte-Tan *et al.* l 2007	Fisheries benefits only

Habitat	Location	Data type	Habitat value ($ ha^{-1} yr^{-1})	Rehabilitation cost ($ ha^{-1})	Source	Comments
Seagrass	Italy	Primary	6822		Scanu *et al.* 2022	Value commercial fishery
Seagrass	Indonesia	Primary	149		Unsworth et al 2010	
Seagrass	Australia	Primary		2,026,895	Walker *et al.* 2003	
Seagrass	Australia	Primary	3,359		Watson *et al.* 1993	Commercial prawn harvest only
Seagrass	Global	Review		132,770	Bayraktarov *et al.* 2016	
Seagrass	Global	Review		17,000-1,258,000	Grabowski *et al.* 2012	
Seagrass	Global	Review	11,907		Spurgeon 1992	
Seagrass	Global	Review	3,067	24,000-1,824,013	Spurgeon 1999	
Swamps & floodplains	Global	Review	115.8		Costanza *et al.* 1997	Mean value from global review
Unvegetated sublittoral	USA	Primary	206		O'Higgins *et al.* 2010	Base value for comparison in marine environment

Table 20 Willingness to pay for habitat rehabilitation relating to fisheries improvement. All values converted to AUD 2021.

Habitat type	Country	Valuation method	WTP (2021 AUD)	Sources	Comments
Mangrove	Bangladesh	CV	12.34 per ha	Rahman *et al.* 2018	
Oyster reefs	USA	MV, BT	24.29 per kg	Lai *et al.* 2020	
Oyster reefs	Australia	TC, MV	interstate - 97 per day Local - 2.96 per trip	Rogers *et al.* 2018	
Oyster reefs	Australia	CV, TC, MV	0.15 per ha for healthy reef (env benefit)	Rogers *et al.* 2018	Fisheries and environmental benefits
Oyster reefs	USA	MV, TC	24.53 per trip	Grabowski *et al.* 2011	
Oyster reefs	USA	TC, RUM	40.81 per trip	Hicks et al 2004	Recreational value with stock increase (productivity increase)
Seagrass	Australia	RUM	20.47 per trip	Huang *et al.* 2020	
Seagrass	Australia	RUM, TC	88.40 per trip	Huang *et al.* 2020	
Rivers	Global	CV, DCE	6.13-336.91 p.a.	Bergstrom & Loomis 2017	Results from all countries
Rivers	USA and Europe	CV, DCE	166.89-336.91 p.a.	Bergstrom & Loomis 2017	Only USA and Europe
Rivers	Europe	CV	52.64-168.45 p.a.	Ayres *et al.* 2014	User WTP
Rivers	USA	DCE	124.35 p.a.	Brouwer & Sheremet 2017	User WTP
Rivers	USA	CV	277.19 p.a.	Loomis 1996	WTP per household per year for 10 yr
Rivers	USA	CV	145.42-179.92 p.a.	Bell *et al.* 2003	WTP per household per year

Habitat type	Country	Valuation method	WTP (2021 AUD)	Sources	Comments
Rivers	USA	CV	7.32-123.24 p.a.	Bell *et al.* 2003	WTP per household per year
Rivers	USA	CV	61.64-355.75 p.a.	Olsen *et al.* 1991	WTP per household per year
Rivers	USA	CV	109.80-308.22 p.a.	Layton *et al.* 1999	WTP per household per year
Rivers	Sweden	BT	373.67-784.63 p.a.	Hnin 2017	WTP per person per year
Rivers	UK	CV	2426.75 p.a.	Thomas & Blakemore 2007	WTP per person over 10 years
Rivers	UK	CV	172.79 p.a.	Thomas & Blakemore 2007	WTP per person over 10 years
Rivers	UK	CV	172.79 p.a.	Thomas and Blakemore 2007	WTP per person over 10 years

Wetland ecosystems are important to fisheries, but estimating the economic value of these services requires some quantitative estimate of the linkage between ecosystems and the fisheries they support (Abrantes *et al.* 2015). Recent efforts to build a business case in support of wetland habitat rehabilitation are reliant on demonstrable habitat–fishery linkages (Sheaves *et al.* 2014, Creighton *et al.* 2015, Taylor 2016, Raoult *et al.* 2018). Defining habitat–fishery relationships in a quantitative fashion is essential to establish the potential economic outcomes that can result from habitat repair through trophic support of commercial and recreational fisheries productivity. The number of studies investigating this field and the sophistication of their approaches is increasing, providing managers with a clearer understanding on which to base investment decisions.

It should be noted that the marginal values of rehabilitation in different habitat types may vary because of biological and/or social factors. The functional role of enhanced habitat for fish is probably influenced by the amount of existing habitat available for finfish and exploited crustaceans in a given location. Therefore, the marginal value of each unit of restored habitat may decrease as habitat is restored in the system, especially if large restoration projects or efforts in areas with large amounts of existing habitat result in fishery production for fish and crustaceans becoming limited by factors other than habitat availability. Similarly, reductions in recreational catch caused by habitat loss might also affect willingness to pay (WTP) for a day of recreation. However, without original economic studies for a given case study, the variability of WTP with changes in the area of habitat, remain unknown (O'Higgins *et al.* 2010). Few studies have presented such data on marginal benefits, and it is typically assumed that the full benefit per unit of habitat rehabilitated is gained.

Habitat rehabilitation can deliver substantial economic benefits, not just to fisheries. Few studies appear to report on the economic outputs resulting from habitat rehabilitation projects aimed at fisheries enhancement, The following example of the economic returns from national investment in aquatic habitat rehabilitation in the USA highlights the potential scale of return, and also how different metrics provide different results. Samonte *et al.* (2017) found NOAA's 2009 $216.1 million investment in stream, wetland and coastal habitat restoration restored 10,354 ha of habitat, opened 1,090 km of stream for fish passage and removed 393,171 tonnes of debris from wetland and upland habitats. Rehabilitation activities directly and indirectly supported 2,280 full and part-time jobs. Labour income from jobs directly or indirectly supported from the NOAA projects was $163.5 million. In total, the project's contributed $200.7 million to the U.S. economy in terms of value added. Counting direct, indirect, and induced impacts, the investment of $216.1 million resulted in $365.5 million in total economic output. Of these benefits, increased outputs due to fishery enhancement were only one small component, and if viewed in isolation, would not have produced economically feasible outcomes in this instance.

The economic outcomes from rehabilitation projects are typically very scenario specific. Some projects can produce positive economic outputs and outcomes, whilst others are never likely to provide a positive economic return. Following on from the example above, Table 21 provides a case study of three NOAA funded projects exemplifying this. The short-term economic outputs from the three projects are all greater than investment costs, but the economic outcomes from ecosystem services had mixed results, depending on whether the upper or lower bounds were considered Speers *et al.* (2015). The range of values presented reflect the inherent uncertainty often found in estimations of economic value, particularly non-market benefit valuation. Economic output and ecosystem total present value are complementary measures of value-added from the one-time spending on rehabilitation. Notably, the upper bound estimates of ecosystem service benefits alone often far exceed construction costs. For two of the three projects, ecosystem BCR indicate that, by enhancing and restoring ecosystems, restoration spending generates highly favourable gross returns (Samonte *et al* 2017). Short-term economic output, on the other hand, provides smaller return given the same construction cost. Not accounting for ecosystem benefits may, therefore, lead to incomplete conclusions about restoration benefit-to-cost ratios. Unfortunately, few habitat rehabilitation studies associated with fisheries improvement report on economic outputs, instead focussing on the longer-term economic outcomes from improved ecosystem service benefits.

Table 21 Economic benefits of three coastal restoration projects in the USA (USD 2013). From Speers *et al.* (2015). TPV = Total present value.

Site	Funding	Economic output	Ecosystem TPV	Ecosystem BCR
San Francisco Bay Salt Ponds	$8.27 million	$8.07 million	$6.89 - $220 million	0.8:1 to 27:1
Virginia Seaside Bays	$2.35 million	$2.57 million	$34.9 - $84.8 million	14:1 to 36:1
Mobile Bay, Alabama	$3.18 million	$3.46 million	$183,000 - $337,000	0.06 to 0.1

As noted in the other Chapters of this review, detailed socio-economic data and analysis is generally lacking for fisheries habitat rehabilitation projects. Many of the cost estimates in the literature are not readily comparable because the cost components of restoration included vary greatly and lots are often ignored. The documented expenditure on habitat rehabilitation projects in the literature is often dictated by available budgets. Many of the costs documented in the available literature provide little detail as to exactly what they include. The costs that are most often recorded relate only to the construction and/or financial costs, which are relatively easily identified, or they may represent a particular fund made available for the rehabilitation/creation. Invariably there will be other unrecorded costs such as staff time, facilities and materials provided by numerous organisations involved, and any off-site scheme impacts including those incurred by the donor site (Spurgeon 1999).

4.2.2.1 Mangroves

Mangroves have been globally identified as an important habitat that contributes to the productivity and value of many coastal, offshore and inland fisheries (Table 19). For example, mangrove-related fish and crab species account for 32% of the small-scale fisheries landings in the Gulf of California, with an estimated annual value of \$58,104 ha^{-1} of mangrove fringe (Aburto-Oropeza *et al.* 2008). Similarly, 79% of surveyed households in the Sundarban Mangrove Reserve, Bangladesh rely on various mangrove-supported fisheries as part of their year-round income, providing an estimated habitat value of \$1,550 ha^{-1} (Rahman *et al.* 2018). In Australia, Janes *et al.* (2020) found mangroves supported 19,000 more fish, equivalent to 265 kg^{-1} ha^{-1} y^{-1}, compared to unvegetated habitat. The highest biomass and economic value originated from larger, longer-lived fish that are regularly targeted by fisheries (*e.g.* breams and mullets).

The loss of mangrove habitat devalues fisheries and provides insight into the potential benefits from habitat rehabilitation. Several studies have applied the production-function method to value the role of mangrove systems as nursery and breeding habitat for fisheries in Thailand. Deforestation of mangroves was estimated to reduce fisheries productivity by \$571 ha^{-1} y^{-1} (Barbier 2003) with losses in the shellfish fishery twenty times higher than for demersal fisheries. In another study, Barbier (2007) used a dynamic model to refine estimates, calculating that the NPV (10% discount, 9 year period) of mangroves as breeding and nursery habitat in support of artisanal fisheries ranged from \$1,600 to \$2,229 ha^{-1}. The results clearly demonstrated how loss of mangrove habitat devalued fisheries and provides insight into the potential benefits that could be achieved from habitat rehabilitation.

Several studies have estimated the fishery value associated with mangroves in Australia. Morton (1990) valued the mangroves of Moreton Bay, in southern Queensland, based on the market value of fish caught. The study estimated the fishery value of mangroves at \$20,092 ha^{-1} based on the market value of the fish caught (not taking into account invertebrates or juvenile fish of commercially important species). In NSW, the importance and value of mangrove systems to the commercial school prawn fishery was estimated to be between \$60 ha^{-1} yr^{-1} and \$994 ha^{-1} yr^{-1}.

The median fisheries value of mangrove habitat from the studies reviewed in this report (Table 19) was \$1,233 ha^{-1}. This may be slightly low for Australia due to prevalence of data from developing countries. Su *et al.* (2021) reported the same median value in their meta-analysis, but found the mean to be \$7,780 ± 5,119 ha^{-1} yr^{-1}, indicating that there were several higher values which came from developed countries. The annual mangrove associated fishery value estimated for Moreton Bay (\$20,092 ha^{-1} yr^{-1}) from Morton (1990) is therefore likely to be most indicative for Australia.

Techniques for the rehabilitation of mangrove habitats are well-developed and can be undertaken over extensive areas. This provides potential to generate large-scale impacts for fisheries enhancement at the local or population scales. A range of techniques have been used, but the main two approaches are rehabilitation of hydrological conditions and direct planting. Direct propagule dibbing is a new approach that may bring down rehabilitation costs significantly, but its use is still being evaluated (Chowdhury *et al.* 2018). The costs associated with rehabilitation projects have varied greatly from \$139 ha^{-1} to \$1.63 million ha^{-1}, depending on the nature of the activities undertaken (Table 19).

In a meta-analysis of the cost and feasibility of marine habitat restoration, Bayraktarov *et al.* (2016) reported the median rehabilitation cost of 59 studies from developed countries was \$48,469 ha^{-1}. More recent meta-analyses by Taillardat *et al.* (2020) and Su *et al.* (2021) reported total mangrove restoration costs ranged \$23 to 371,327 ha^{-1} with median values of \$5,431 ha^{-1} and \$1,616 ha^{-1}, respectively. However, these studies incorporated many restoration projects from developing countries. Taillardat *et al.* (2020) noted that there was a two order of magnitude difference between the median costs of mangrove restoration in developed (\$125,408 ha^{-1}) and developing (\$1,230 ha^{-1}) countries. This may be explained by the fact that mangrove restoration projects in Southeast Asian countries often do not involve mechanical earthworks; instead, earthworks are carried out by the local community using hand tools. Therefore, the median value from Bayraktarov *et al.* (2016) may be more applicable to indicative rehabilitation costs in Australia. This is supported by the results from a study by Melbourne Water (2013) examining mangrove rehabilitation in Western Port Bay, where mean mangrove rehabilitation costs by paid employees were \$63,325 ha^{-1}. Volunteer groups play a role in many mangrove restoration projects in Australia, and their involvement would likely reduce the restoration costs similar to the median cost estimated by Bayraktarov *et al.* (2016).

There have been several documented examples where the rehabilitation of mangrove systems has contributed significant economic benefits to associated fisheries. In Java, Indonesia, the mangrove-associated finfish fishery increased from practically zero to becoming a profitable livelihood option, over a 3 year inshore period following mangrove restoration (Debrot *et al.* 2022). The fishery increase occurred after 8.5% of a highly degraded 419 ha mangrove area had been rehabilitated and also resulted in an increase in fishers operating in the rehabilitated area.

Commercial catch data from Gujarat, India, also showed that planted mangroves significantly increased the catch of mangrove-dependent fish, crustaceans and mollusc species in both inshore and offshore fisheries (Das 2022). Since mangrove restoration commenced, the associated inshore and offshore fisheries annual catch have increased by \$15% or \$9,598 ha^{-1} yr^{-1} planted mangroves. This is a significant value, despite the contribution of young, planted mangroves to the fishery being only 22% that from mature natural mangroves. Even considering this, inputs from planted mangroves to Gujarat's fishery sector were valued at \$781 million yr^{-1}.

Socioeconomic analysis of the costs and benefits accruing to coastal communities in the Philippines participating in a successful cooperative mangrove replanting initiative showed direct economic benefits of \$1,074-4,412 $ha^{-1}.yr^{-1}$, including contribution from timber products, mangrove fisheries, and adjacent catches of mangrove-associated species (Walton 2006). Importantly, for such community-based initiatives, this income accrued directly to community members and a survey of attitudes of fishers indicated that they valued the benefits of mangrove restoration very highly. It was also demonstrated that rehabilitating mangrove systems could successfully help in restoring an economically important crab species to levels expected from natural mangrove stands (Walton *et al.* 2007).

Barbier (2000) estimated the annual benefits of mangrove associated fisheries from rehabilitation activities in Thailand. At a large scale (800 ha) mangrove restoration in project in Surat Thani converting abandoned shrimp aquaculture ponds back to their former condition as mangrove forests successful restoration costs using just hydrologic restoration was \$452 ha^{-1}, or \$1,581 ha^{-1} if planting mangroves was also used. Barbier (2000) then estimated that the economic loss to the Gulf of Thailand fisheries due to removal of 1,200 ha of mangroves was \$225,870 yr^{-1}. Based on the restoration costs from the Surat Thani project, it would take about three years and cost between \$542,000-1,897,000 to restore the mangroves. Without factoring in a discount rate, these figures indicate that the cost of restoration would be recovered in restored fisheries values within 2.4-8.4 years, and then would continue to be generated without additional costs in perpetuity.

Only two studies were identified that included detailed benefit cost analyses, both which generated positive economic outcomes (Table 22). A meta-analysis by Su *et al.* (2021) found that mangrove restoration was cost-effective, with positive benefit cost ratios under variable discount rates when a range of ecosystem services are included. The median economic benefit to fisheries from restoring mangroves was \$1,233 ha^{-1} yr^{-1}, whilst the median mangrove restoration cost was \$1,616 ha^{-1}. Mangrove restoration BCR ranged from 10.50 to 6.83 (for discount rates of -2 to 8%) when summing the total benefits from each ecosystem service, and from 3.36 to 2.19 when using the mean value of the estimated total economic benefits. Su *et al.*(2021) postulated that the relatively low economic benefits accruing from mangrove restoration might be due to the immaturity and lower diversity of restored mangroves. In the economic studies they reviewed, the reported age of the restored trees was below 20 years. It has been suggested that younger mangroves are less productive compared to mature systems, which may initially limit fishery productivity and take up to 40 years to reach optimum yields (Hutchison *et al.* 2014, Lahjie *et al.* 2019).

In Vietnam, large areas of mangroves have been converted to agriculture, causing ecological disturbance and enhancing instability in the coastal physical environment (Hong and San 1993). There is a growing trend for local communities to reverse the decline by undertaking mangrove restoration or rehabilitation. In the Nam Dinh Province, the economic value of such activities was assessed by Tri *et al.* (1998), with particular respect to coastal erosion control and storm protection. However, the analysis also included assessment of the economic impacts on local commercial fisheries. It was estimated that it would cost \$133 ha^{-1} to restore mangroves, value-adding \$166 ha^{-1} yr^{-1} to the local fishery. The present value of costs (6%, 25 years) would be \$667 ha^{-1}, and the present value of fishery benefits \$1,966 ha^{-1} over the same period. The NPV was \$1,299 ha^{-1}, with a BCR of 2.95. Therefore the economic benefits accrued in the fishery sector would more than offset the restoration costs and the project would be economically feasible purely from a fisheries enhancement perspective.

Although mangrove restoration can deliver significant benefits to fisheries, the cost of restoration means that rehabilitation projects may sometimes not be economically feasible based on fisheries benefits alone. However, comparing the calculated fisheries benefit (\$20,092 ha^{-1} y^{-1}) of Queensland mangroves from Morton (1990), with the median cost of mangrove restoration (\$48,469 ha^{-1}) reported by Bayraktarov *et al.* (2016), would generate NPV (30 years, 5% discount rate, 10% progressive annual increase in benefits in first 10 years) of \$184,359 ha^{-1} with a BCR of 4.8. These results suggest that mangrove restoration has the potential to be an economically viable fisheries enhancement tool for managers under the right circumstances.

Table 22 Examples of economic value results from cost benefit analyses on habitat rehabilitation for fisheries enhancement. Values in AUD 2021 for easier comparison.

Habitat type	Location	Restoration area (ha)	Analysis method	Fishery type	Equivalent annual value (AUD 2021)	NPV (AUD 2021)	BCR	Sources	Comments
Coastal wetlands	USA		CVM	R		86,263,779		Bergstrom *et al.* 1990	Fisheries value only
Coastal wetlands	USA		MV	C		827 ha^{-1}		Lynne *et al.* 1981	Fisheries value only
Coral reef	Philippines		MV	C	$583,000	442,000	4.13	White *et al.* 2009	Commercial fishery benefit from rehabilitation
Freshwater marsh	Canada		TC	R	13,466			van Vuuren & Roy 1993	4%, 50 years
Mangrove	Vietnam		MV			1,362-3,321 ha^{-1}	4.7-5.7	Ledoux *et al.* 2002	Includes value of timber and honey production
Mangrove	Bangladesh		MV	C	487,982,443			Rahman *et al.* 2018	Fisheries value only
Mangrove	Global review		MV	C			6.83-10.50	Su *et al.* 2021	Review
Mangrove	Vietnam	4.8	MV	C		1,299	2.95	Tri *et al.* 1998	6%, 25 yr, Fisheries value only
Oyster reefs	USA	1,045	MV, BT	C, R	27,551,818			Lai *et al.* 2020	Commercial harvest and angler WTP
Oyster reefs	USA	8.9	MV, CVM	C,R	59,036	-1,078,629	0.61	Speers *et al.* 2015	4%, 40 yr, Commercial and recreational fisheries value only. No oyster harvest'
Oyster reefs	USA	23	TC	R	890,525	8,544,397	1.23	Carlton *et al.* 2016	Recreational use and guiding only
Oyster reefs	USA	221		C, R		18,989,608	1.61	Callihan *et al.* 2016	3% disc. 15 yr, Fisheries benefits only. Commercial activities include oyster harvest
Oyster reefs	USA	221		C, R		45,942,601	2.73	Callihan *et al.* 2016	3% disc. 25 yr. Fisheries benefits only. Commercial activities include oyster harvest
Oyster reefs	USA	18.6	MV, TC	C, R	243,571-1,578,711	-2,238,194-21,278,489	0.52-5.55	Grabowski *et al.* 2011	3%, 25 yr, Fisheries value only
Oyster reefs	USA	765	TC, RUM	R	1,747,067	-42,322,757	0.45	Hicks *et al.* 2004	Recreational value no stock increase (aggregation effects)
Oyster reefs	USA	765	TC, RUM	R	13,972,176	197,294,772	3.58	Hicks *et al.* 2004	Recreational value with stock increase (productivity increase)

Habitat type	Location	Restoration area (ha)	Analysis method	Fishery type	Equivalent annual value (AUD 2021)	NPV (AUD 2021)	BCR	Sources	Comments
Oyster reefs	USA	24	CVM, MV	C, R		6,962,871	2.3	Kroeger 2012	50 yr, Fish, crabs and oyster harvest. 50 yr NPV
Oyster reefs	Australia	16	TC, MV	C, R		4,087,168	1.15	Rogers *et al.* 2018	Considering benefits accrued only to rec, commercial and oyster suppliers only
Oyster reefs	Australia	16	CVM, TC, MV	C, R		9,584,283	2.6	Rogers *et al.* 2018	Fisheries and Environmental benefits
Oyster reefs	USA	22				947,000		Shephard *et al.* 2016	Generated economic output of $1.75 million
Saltmarsh	USA	613	BT	C, R	102,575			Speers *et al.* 2015	3%, 40 yr, Commercial and recreational fisheries value only
Seagrass	Australia	+10%	RUM	R		6,406,486		Huang *et al.* 2020	
Seagrass	Australia	+30%	RUM, TC	R		22,732,691		Huang *et al.* 2020	
Seagrass	Australia		CVM	R		-93,779	0.5	Rogers *et al.* 2019	
Seagrass	Australia		CVM	R		53,980	2.5	Rogers *et al.* 2019	
Rivers	USA		MV, CVM	C, R		-417,580,557	0.02	Bellas & Kosnik 2019	7% 100 years
Rivers	USA		BT	R		32,406,129	1.81	Cook & Becker 2016	6% 20 years
Rivers	USA		BT	R		4,545,656	1.21	Cook & Becker 2016	3% 20 years
Rivers	Sweden		BT	R		14,542,902	2.57	Hnin 2017	3.5%, 10 years
Rivers	USA		DCE	R		542,141	11.75	Holmes *et al.* 2004	3.5%, 10 years
Rivers	USA		TC	C, R		-12,485,393		Loomis 1989	30 yr
Rivers	England			C, R	2,692,630	20,464,674	6.7	Vardakoulias & Arnold 2015	3.5%, 10 years, fisheries only
Rivers	UK	2.7 km	CVM	R		1,674,311	28.6	Thomas & Blakemore 2007	6%, 10 years

Habitat type	Location	Restoration area (ha)	Analysis method	Fishery type	Equivalent annual value (AUD 2021)	NPV (AUD 2021)	BCR	Sources	Comments
Rivers	UK	20 km	CVM	R		1,285,707	3.9	Thomas & Blakemore 2007	6%, 10 years
Rivers	UK		CVM	R		-2,147,547	0.5	Thomas & Blakemore 2007	6%, 10 years

4.2.2.2 Seagrass

Attempts at seagrass ecosystem service valuation have used a variety of approaches., which by and large have been location specific, and covered only a partial list of values. They typically reflect the nature of the ecosystem service provided in the area (Dewsbury *et al.* 2016). There have been many small-scale restoration trials that have shown success. However, the challenge remains to translate small-scale success into large-scale restoration programs (van Katwijk *et al.* 2016).

In this section, only ecosystem service values directly attributed to recreational and commercial fisheries productivity have been included because these will assist managers to decide whether the resultant benefits from seagrass restoration can justify investment from a fishery perspective. The economic value of fisheries production in seagrass meadows varies greatly, ranging from as low as \$149 ha^{-1} y^{-1} up to \$35,619 ha^{-1} y^{-1} (Table 19). Reviews by Spurgeon (1992) and Spurgeon (1999) listed the mean economic values as \$11,907 and \$3,067 ha^{-1} y^{-1}, respectively. In the current review, the median value from all included literature was \$3,212 ha^{-1} y^{-1}.

In Australia, slightly higher values have been reported. Across southern Australia the fisheries value of seagrasses has been estimated at \$35,619 ha^{-1} y^{-1} using enhancement estimates related to nursery habitat availability (Blandon and zu Ermgassen 2014a,b, see Case Study). Other studies in Australia have valued the fisheries contribution of seagrass between \$1,996 and \$21,880 ha^{-1} y^{-1} (Table 21). Watson *et al.* (1993) estimated the mean potential economic value of \$3,359 ha^{-1} y^{-1} (range = \$365-3,750 ha^{-1} y^{-1}) from prawn harvest derived from the standing stock of juveniles in seagrass in northern Australia. McArthur and Boland (2006) found the fish, shrimp, and crab yield from seagrass in southern Australia was valued at \$1,996 ha^{-1} y^{-1}. Janes *et al.* (2020) estimated 99% of the economic enhancement of fisheries identified in coastal systems originated from seagrass ecosystems. However, significant spatial differences were observed in the economic values: \$63,900 ha^{-1} y^{-1} in New South Wales, \$1,586 ha^{-1} y^{-1} in Victoria, and \$154 ha^{-1} y^{-1} in South Australia. The average value for seagrass beds across Australia was estimated at 21,880 ha^{-1} y^{-1}.

The reported restoration costs for seagrass are also highly variable, ranging from \$17,000 ha^{-1} up to \$1.82 million ha^{-1} (Table 19). Bayraktarov *et al.* (2016) reviewed 64 seagrass published seagrass restoration studies from developed countries and found the median rehabilitation cost was \$132,770 ha^{-1}. Rogers *et al.* (2019) recently estimated that the average seagrass restoration costs in Western Australia for work undertaken by professionals would be \$194,701 ha^{-1} for replanting and \$38,993 ha^{-1} for reseeding.

Several studies have found rehabilitating seagrass meadows for fisheries benefits to be economically feasible (Case Study, Table 22). Anderson (1989) estimated that restoration of the seagrass in Chesapeake Bay, USA, would deliver substantial economic benefits to the local hard-shell blue crab fishery. It was estimated that an increase in producer surplus of \$5.08 million and \$6.83 million consumer surplus would be generated from the restoration of 1,700 ha in the bay. This would equate to a value-add of \$2,985 ha^{-1} for the commercial fishers and \$4,021 ha^{-1} for consumers, yielding long-run annual net benefits of \$12.04 million or \$7,006 ha^{-1} yr^{-1} for the fishery.

Case Study 11: Seagrass restoration for fisheries enhancement in southern Australia

A review by Blandon and zu Ermgassen (2014a) from southern Australia provided a significant contribution to our understanding of seagrass-fishery relationships, combining quantifiable, large-scale ecological data with economic analysis. The enhancement of juvenile fish abundance provided by the presence of seagrass habitats was estimated by conducting a meta-analysis of juvenile fish abundance in seagrass vs. unvegetated sites in southern Australia. Each hectare of seagrass restored in southern Australia was predicted to enhance commercial fish species by a total of 9,800 kg ha^{-1} y^{-1}. The economic value of the enhancement was estimated at \$31,650 ha^{-1} y^{-1} once all fish are fully recruited to the habitat (>26 years after restoration). This figure is comparable with other previous valuations on seagrass fisheries productivity from around the world (Table 19). The cost effectiveness of restoration on the basis of commercial fish recruitment enhancement suggested that a low cost seeding approach to restoration, costing \$10,000 ha^{-1} (Wear *et al.* 2009), would have a payback time of less than 3 years (at a 3% discount rate). The highest cited cost of \$1,308,284 ha^{-1} (Ganassin and Gibbs 2008), however, had an infinite payback time if only enhancement of the twelve commercial fish species considered was taken into account. A maximum restoration cost of \$901,000 ha^{-1} could be justified on the basis of the recruitment enhancement of these commercial fish species alone. However, Tarwhine provides a large proportion of the economic enhancement, and if removed from the calculations, the payback period rises to 11 years for the same rehabilitation cost of \$10,000 ha^{-1} and the maximum justifiable rehabilitation cost falls to \$58,000 ha^{-1}. This latter value is likely to be more broadly applicable and better compares with the results from other studies (Table 22).

Huang *et al.* (2020) quantitatively predicted the recreational fishery welfare gains from of seagrass rehabilitation in Port Phillip Bay and Western Port, Victoria, which increased coverage by 10% or 30%. The economic benefits for anglers varied widely across space due to the heterogeneous coverage of seagrass in fishing locations, ranging from near-zero to \$20.47 per trip corresponding to 10% increase in seagrass cover and to \$88.40 per trip for a 30% increase in seagrass coverage in locations of existing high seagrass coverage. A 10% increase in seagrass area across the two bays would see growth in economic benefit to recreational fishing of at least \$6.4 million per year, while a 30% increase could add over \$22.7 million each year. However, despite these high potential economic outcomes, it was concluded managers will need to target ecosystem rehabilitation in high value locations given the high cost of seagrass rehabilitation.

Rogers *et al.* (2019) conducted a cost benefit analysis to understand the economics of seagrass restoration options for two projects in Western Australia. The costs associated with seagrass restoration by professional and volunteers for two restoration approaches were compared. The benefits included in calculations were carbon sequestration and non-market values. Whilst not directly measuring fisheries benefit, recreational fisheries benefits were deemed to be captured through the non-market benefits which were estimated from annual WTP values for households. Benefits arising for commercial fisheries were not considered. The key results from the analysis were that replanting seagrass methods relying on professional staff were not economically viable (BCR = 0.5), whilst reseeding methods were always economically viable (BCR > 2.5) and had a greater capacity to manage the risks of project failure than replanting methods. Projects using volunteer-based labour sources delivered larger net benefits than those using professional labour only, and net benefits were largest for projects with larger spatial extents. With the exclusion of the professional-labour replanting scenarios, where costs exceeded benefits, all scenarios had positive NPV. The net benefits ranged from roughly \$42,000 ha^{-1} for a replanted volunteer-based plot, to over \$75,000 ha^{-1} for a reseeded volunteer-based plot. Where positive net benefits were projected, the payback period ranged from 8 to 17 years. It should be noted that the proportion of total benefits accrued for recreational fisheries was not stated and that if only fisheries benefits were considered, the NPV would likely be closer to neutral or negative.

Although seagrass restoration can deliver significant benefits to fisheries, the high cost of restoration means that rehabilitation projects are sometimes not economically feasible based on fisheries benefits alone. However, comparing the recently calculated mean fisheries benefit (\$21,880 ha^{-1} y^{-1}) of seagrass across southern Australia from Janes *et al.*(2020) with the mean cost of seagrass restoration (\$132,770 ha^{-1}) reported by Bayraktarov *et al.* (2016) or the recent results from Western Australia (\$139,913 ha^{-1}, Rogers *et al.* 2019), would generate NPV (30 years, 5% discount rate, 1 year delay for benefits) of \$183,000 ha^{-1} and \$176,000 ha^{-1}, and BCR of 2.4 and 2.3 respectively. These results suggest that seagrass restoration has the potential to be an economically viable fisheries enhancement tool for managers under the right circumstances.

4.2.2.3 Shellfish reefs

Fisheries enhancement values are often listed as one of the driving forces behind shellfish reef rehabilitation or restoration. Shellfish reefs have been well demonstrated to provide significant economic value to commercial and recreational fisheries (Table 19). The value of fish produced by a unit of shellfish reef varies as a function of many ecological and economic factors as well as how these species are managed (Grabowski *et al.* 2012). Several studies have indicated that restored oyster reefs can enhance the overall carrying capacity of exploitable fish and augment fish populations. (*e.g.* Peterson *et al.* 2003 Gilby *et al.* 2020). The density of harvestable fish has been reported to increase by up to 16.4 times, whilst fish distributions across the broader landscape did not change, suggesting enhanced production rather than aggregation.

Most of the reported studies which included economic valuation of shellfish reef rehabilitation have been conducted on oyster reefs. The reported fishery value for oyster reefs in the reviewed literature varied widely, ranging from \$2,020 ha^{-1} up to \$380,289 ha^{-1}, with a median value of \$13,077 ha^{-1} (Table 19). Grabowski *et al.* (2012) reviewed the economic value of oyster reef services, excluding oyster harvesting, conservatively estimating that it was \$5,010 ha^{-1} yr^{-1} for commercial fish and invertebrates, and between \$9,723 ha^{-1} yr^{-1} and \$176,000 ha^{-1} yr^{-1} when all ecosystem services were considered. Grabowski and Peterson (2007) found a slightly higher value of augmented commercial fish production for 13 exploited species that were enhanced by oyster reef habitat. The annual value-add of landings was estimated at \$7,160 ha^{-1} yr^{-1} with a value-added present value (50 yr, 3% discount rate) of \$189,753 ha^{-1} for each hectare of restored oyster reef. Again, the commercial fisheries value was only a small component of the average annual value of all services provided by restored and protected oyster reefs, depending on where the restored reef was located and the suite of ecosystem services that the restored reef provides.

Rogers *et al.* (2018) provided the only identified fishery value calculated for an Australian shellfish reef restoration project. Construction of Windara Reef was considered to have no marginal benefit for the commercial fishery sector, but would value-add \$29,014 ha^{-1} yr^{-1} for Charter operators and the recreational sector. A large component (92%) of this value was driven by the increase in interstate recreational angler visitation.

It should be noted that the marginal value of each unit of restored oyster reef may decrease as more reefs are restored in the system (Hanley and Barbier 2009). This is especially the case if large restoration projects occur in areas with substantial amounts of existing reef habitat. Fishery production for exploitable reef fish and invertebrates may become limited by factors other than habitat availability. No studies evaluating this marginal effect were identified.

Shellfish reef restoration is not cheap because typically new hard substrate needs to be deployed to create reef structure for the shellfish to attach onto (Table 19). Not only do substrates vary markedly in their efficacy in promoting oyster reef growth, but they also vary in their monetary and environmental cost. In instances where natural products such as shell and rock are appropriately placed, they generally offer greater return on investment than the use of manufactured products (Hernández *et al.* 2018).

Oyster reef rehabilitation costs have been more frequently reported than fisheries values. Reported project costs in the reviewed literature again ranged widely, from \$100,106 ha^{-1} to as high as \$5,321,623 ha^{-1}, with a median value of \$168,770 ha^{-1}. The review by Grabowski *et al.* (2012) found a similar range of costs (\$63,000-316,000 ha^{-1}). Hernandez *et al.* (2018) reviewed recovery efforts for eastern oysters in the USA and found oyster reef restoration construction costs ranged widely, from a low of \$5,610 ha^{-1} to a high of \$3,197,063 ha^{-1}, with a mean of \$439,889 ha^{-1}. All studies were carried out in developed countries so the results should be reasonably indicative of construction costs in Australia. The variation observed in project costs suggests that site-specific factors play a key role in the cost-effectiveness of using shellfish reef restoration. Rogers *et al.* (2018) reported costs of \$223,933 ha^{-1} for the construction of Windara shellfish restoration reef in South Australia (see Case Study 12). This particular reef was located offshore in Gulf St Vincent, which may account for the slightly higher installation costs.

Restoration of shellfish reefs in Australia commonly use recycled cleaned bivalve shells to create settlement substrate and reef structure. However, translocation and reuse of these shells poses biosecurity risks for the transfer and spread of disease (*e.g.* QX disease) and marine pests (Diggles 2020). If disease or pests are transferred, the detrimental impacts on wild and cultured shellfish stock may be expensive. The potential financial costs from lost production, clean-up and restoration may need to be considered in planning decisions. Standard operating procedures have been developed in Australia to minimise these risks and should be followed for all projects using recycled shells.

The literature search identified 11 studies which included some economic analysis of the costs and benefits associated with oyster reef rehabilitation with respect to fisheries enhancement (Table 22). The scale of oyster reef rehabilitation projects ranged from small 1 ha trial sites, through to entire estuarine bay systems covering 765 ha. The NPV for oyster reef rehabilitation based solely on the enhanced commercial and recreational fisheries values were positive in 8 of the 11 studies. When other ecosystem benefits were included, all studies had positive NPV, indicating that they would be economically feasible. The mean BCR was 1.75 ± 1.17 (SD), suggesting that overall, oyster reef restoration is likely to deliver positive economic outcomes based on fishery enhancement.

Case Study 12: Windara Reef

The Windara Reef is the largest (20 ha) subtidal shellfish reef attempt undertaken in Australia. The total cost of reef construction has been estimated at $3.4 million (AUD 2017). Rogers *et al.* (2018) undertook a cost-benefit analysis and integrated economic assessment of the 16 ha stage two phase of the project. This analysis included the tangible, market-based outcomes and also the intangible, non-market social and environmental outcomes of the project. Construction of Windara Reef was considered to have no marginal benefit for the commercial fishery sector, but would value-add $25,583 ha^{-1} yr^{-1} to the recreational fishery and $1,950 ha^{-1} yr^{-1} to the Charter boat fishery. Environmental benefits constituted the greatest value-add from reef construction at $326,000 ha^{-1} yr^{-1}.

Construction of Windara Reef was predicted to likely be economically feasible from a fishery enhancement perspective. The NPV to fisheries over a 30 year period (7% discount) was estimated to be $3,802,892, with a BCR of 1.14. The key benefit driving project viability was the additional spend that could be generated for the tourism sector by interstate recreational fishers. These recreational benefits depend critically on assumptions about whether the new reef will create additional fishing trips, or whether there will be substitution of effort from elsewhere, but with increased satisfaction for those trips.

If the estimated values for environmental benefits were also included, the project demonstrated between two to four times return on investment and generated net benefits of between $4 million to $10 million, depending on the discount rate applied and duration of operating costs. Across the modelled scenarios in the sensitivity analysis, the NPV of the project ranged from -$2 million to $5.0 million, and the BCR from 0.4 to 15.7. Only the project scenarios that assumed the lower bound values for all benefits at a discount rate of either 7% or 10% resulted in a negative outcome. In all other instances, the project was economically viable, and demonstrated substantial capacity to absorb the risk of total project failure. It was interesting to note that extending the period of monitoring from 2 to 10 years did not alter the viability of the project. This suggests that the ongoing monitoring of the reef to collect baseline data to inform future restoration investments is worthwhile and that extending the duration of monitoring in other shellfish restoration projects should be considered.

A cost benefit analysis of the oyster reef enhancement program in North Carolina, USA, by Callihan *et al.* (2016), evaluated the economic outcomes from the $30.9 million invested in restoration activities. Improvements in the commercial and recreational fisheries combined with coastal water quality gains produced positive economic benefits of $73.5 million over 15 years, or $125.3 million over 25 years, generating NPVs (3% disc) of $42.6 million and $94.3 million, and BCRs of 2.38 and 4.05, respectively. The economic benefits derived solely from fisheries were also substantial. Oyster reef enhancement was expected to generate benefits for recreational and commercial fisheries of $48.9 million and $84.5 million, over 15 and 25 years, respectively. The BCRs for fisheries enhancement were therefore 1.61 and 2.73. Callihan *et al.* (2016) also estimated annual commercial fishing sales inputs increased a total of $12.6 million during the 6 year reef enhancement period. These were the direct commercial benefits from habitat enhancement activities. This increase rippled through the North Carolina economy and generated total state-wide economic impacts up to $48.7 million in business revenue, 696 jobs and $20.2 million in annual wages and salaries.

In Matagorda Bay, Texas, the 23 ha Half Moon oyster reef was constructed on the remains of a functionally dead historical 200 ha oyster reef (Callihan *et al.* 2016). The restored habitat has generated significant numbers of large oysters and an increase in biodiversity, including more shellfish, small invertebrates, and harvestable fish, to create an increasingly popular hotspot for hundreds of sport anglers across the region (De Santiago *et al.* 2019). A socio-economic assessment by Carlton *et al.* (2016) found anglers and guides fishing the reef reported higher overall trip satisfaction, caught more fish, and increased fishing frequency. Reef construction cost $6.96 million (Dumesil and Pollack 2014) and directly generated 17 jobs over the life of the project. Post-construction the reef has generated substantial positive economic impacts, including 12 additional jobs, $599,246 in annual labour, $890,492 in annual value-added (state GDP) and $1,640,515 annual output (economic activity). The NPV (3% discount, 25 years) is $8.54 million, returning a BCR of 1.23.

Cost-benefit analysis was also used by Grabowski *et al.* (2011) to assess the long-term value that would be derived from restored oyster reef projects in coastal North Carolina, USA, by coupling market and non-market approaches to determine the long-term value of the ecosystem services provided by rehabilitated oyster reefs. The annual commercial fisheries values from 18 ha of restored oyster reef sanctuaries was estimated to be $95,222 present value per year. Recreational value for the enhanced fishery was estimated at a WTP of $24.53 per trip, giving a present value added of between $148,349 and $1,483,489 depending upon the level of increased recreational effort (0.1% or 1.0%). The annual combined fishery value of the enhanced oyster reefs was therefore estimated to be between $243,571 yr^{-1} and $1,578,711 yr^{-1}. Reef construction costs were estimated at $7,955 ha^{-1}, giving a total cost of $4,578,485. Using a 3% discount rate, it was estimated that the restored oyster reefs would at minimum break even in 24 years from the enhanced fisheries values alone. The NPV (3% discount, 25 years) based on fisheries benefits alone range from $-2,238,194 using the minimum recreation present value, to $21,278,489 using the maximum calculated recreational value, with BCRs of between 0.52 and 5.55, respectively. When the minimum values of all of the ecosystem services provided by the restored reefs were taken into consideration, the NPV (3%, 25 years) was $11,305,841, with a BCR of 3.42 and break-even period as little as 9 years.

Hicks *et al.* (2004) estimated the total net economic benefits of oyster reef restoration in the Chesapeake Bay. The estimated cost of restoring 765 ha of oyster reef was $76,556,000. If there was no net increase in the population size of associated targeted fish species, annual net benefits for recreational anglers would increase $1,747,067 through the concentration of fish around the reefs. The NPV (3%, 30 years) was estimated to be $-42,32 million, with a BCR of 0.45. This NPV is equal to just under 50% of the reef restoration project cost over the 30 year period. If the oyster reef restoration not only provided additional reef to fish, but also led to an increase in the fish population, the annual net benefit for anglers would be $13,972,176 with an NPV of $197.29 million and a BCR of 3.58. The costs for reef construction would be recovered in under 5 years. Besides fishery benefits, the non-use benefits were also estimated to be substantial. WTP for constructed oyster reef was estimated to be at least $40.81 per household per year with a median estimate of $237.8 per household per year. Aggregating to the general population, the non-use value of a ten year oyster reef project, consisting of 4,000 ha of oyster sanctuary and 400 ha of artificial reef was estimated to be at least $314.64 million.

In Mobile Bay, Alabama, 1.4 ha of oyster reefs were installed to provide oyster, other shellfish, and fin-fish habitats and create protective coastal breakwaters to provide shoreline stabilization and resiliency. Speers *et al.* (2015) estimated over $3.8 million was invested in reef construction costs, but the annualised benefits to fisheries were only $2,472 yr^{-1} for the commercial fishery and $6,344 for the recreational fishery. This generated estimated total present values of $50,134 and $128,656 yr^{-1}

respectively. The values for other ecosystem services such as protection from coastal erosion, and nitrogen and carbon sequestration, were of a similarly low magnitude, resulting in the project having a total present value $236,894 to $433,747 and very low BCR (0.06 to 0.12), suggesting the project was not economically viable overall.

A variety of shallow coastal ecosystems, including submerged eelgrass meadows and oyster reefs, in Virginia's seaside bays have declined substantially. To address this, $2.8 million was invested to construct 8.9 ha of oyster and plant 40 ha of seagrass to restore their ecosystem functions and services. An economic analysis of the project by Speers *et al.* (2015) estimated that oyster reef establishment would provide $ $43,961 yr^{-1} (4%, 40 years) in annual benefit to commercial fisheries, with a total present value of $1,029,252. For recreational fisheries the annualised value was $4,432 yr^{-1} to $29,248 yr^{-1}, for a total present value of $103,753 to $684,771. The annualised benefits for all fisheries was therefore $37,552 to $45,809, with total present value of $1,133,006 to $1,714,024. NPV and BCR could not be calculated because the investment split between oyster reef construction and seagrass restoration costs was not provided. However, if the total project costs are all allocated to reef restoration, the BCR would be between 0.41 and 0.61.

Kroeger (2012) generated quantitative estimates for the benefits provided by oyster reef restoration in the northern Gulf of Mexico. Two planned oyster reef restoration projects were to install a total length of 5.8 km of oyster reef covering 24 ha. The reefs were expected to generate an additional catch of over 3,129 kg per year of fish and crab species for commercial and recreational fishers. These harvests will generate estimated net benefits of $12,200-$15,500 yr^{-1} in the commercial and $34,800-$42,200 yr^{-1} in the recreational sectors for a total of $47,250-$57,200 yr^{-1}. The higher catch will increase local economic output by an estimated $48,500 p.a. The oyster reef restoration was predicted to be economically feasible on benefit cost grounds. Reef construction costs were estimated to be $5.316 million and over a 50-year timeframe, the NPV of just the fishery enhancement provided by sustainable fisheries harvest (including oysters) was $6.96 million, providing the project a social return on investment of 2.3. If avoided damages from coastal erosion and flooding are considered, the economic rationale for reef restoration becomes even stronger. The economic impacts from reef construction itself would also generate $10.4 million in local output, $3.5 million in earnings and create 88 jobs over a two year period.

Hernandez *et al.* (2018) reviewed recovery efforts for eastern oysters in the USA, but without specific reference to fisheries values. Oyster reef restoration construction costs were found to range widely, from a low of $3826 ha^{-1} to a high of $2,180,361 ha^{-1}, with an average $299,999 ha^{-1}. Assuming average reef construction costs of $439,889 ha^{-1} and ecosystem service values for oyster reefs of $15,140 ha^{-1} yr^{-1}, the return on investment (NPV, 3% discount, 14 years) was positive for only half of the project conducted. Return on investment varied considerably among the types of substrates used and the scale of the project. It was concluded that investment in oyster restoration is more likely to be recouped via ecosystem service benefits when projects are large and in easy-to-access locations, as well as when inexpensive materials are used. No specific mention of the benefits to fisheries were made in this study.

Although shellfish restoration can deliver substantial benefits to fisheries, the cost of restoration is high. This means that rehabilitation projects may sometimes not be economically feasible based on fisheries benefits alone. A benefit cost analysis comparing the median calculated fisheries benefit ($13,077 ha^{-1} y^{-1}) from Grabowski (2011) or the Australian value ($27,533 ha^{-1} y^{-1}) from Rogers *et al*

(2018), with the median cost of shellfish reef restoration ($223,933 ha^{-1}) reported by Rogers *et al.* (2018), would generate NPVs (30 years, 5% discount rate, 10% progressive annual increase in benefits in first 10 years) of $-72,296 ha^{-1} and $95,122 ha^{-1}, with BCRs of 0.68 and 1.42 respectively. These results suggest that shellfish reef restoration has the potential to be an economically viable fisheries enhancement tool for managers under the right circumstances, especially if recreational angling effort is increased.

4.2.2.4 Saltmarsh

The economic value of fisheries production reported in saltmarshes varies greatly. In the reviewed literature it was typically low, ranging from $484 ha^{-1} y^{-1} up to $40,910 ha^{-1} y^{-1} (Table 19). Most studies have only estimated the commercial values for enhanced fishery harvest, and thus a complete value for fisheries productivity is lacking. Bell (1997) was the only study identified which estimated the recreational fishery value of saltmarsh habitat. The recreational fishery value was estimated at $40,910 ha^{-1} y^{-1}, which was significantly higher than commercial values reported in other studies (<$5,200 ha^{-1} y^{-1}). This is most likely due to high value and WTP that recreational fishers place on key estuarine species. In comparison, a review by Spurgeon (1999) listed the mean economic value from commercial fisheries as a mere $533 ha^{-1} y^{-1}. In the current review, the median value from all included literature was $1,095 ha^{-1} y^{-1}. In Australia, commercial fishery production in saltmarsh habitats has also been estimated to be low compared to that of other wetland habitat types, ranging from $484 ha^{-1} y^{-1} in the Hunter River, NSW, up to $5,167 ha^{-1} y^{-1} in the Clarence River, NSW (Taylor *et al.* 2018). In the current review, the median value from Australian studies was also $1,095 ha^{-1} y^{-1}, matching the global results.

The cost of saltmarsh rehabilitation varies widely. In global reviews, Grabowski *et al.* (2012) reported saltmarsh rehabilitation costs to range between $4,000 ha^{-1} and $294,000 ha^{-1}, and Bayraktarov *et al.* (2016) reported a median value of $83,465 ha^{-1}. In Australia, Knight (2018) documented 32 saltmarsh rehabilitation projects from Queensland and New South Wales which contained sufficient details to estimate the rehabilitation costs. Costs ranged from $235 ha^{-1} up to $13,676,471 ha^{-1}, with a mean of $622,054 ± 2,409,964 ha^{-1} (± SD) and median of $83,465 ha^{-1}. The wide range in costs reflected the variety of different techniques used to rehabilitate sites.

Very few studies have conducted cost benefit analyses on saltmarsh rehabilitation projects. Even fewer studies have reported on the results with respect to fisheries enhancement benefits (Table 22).

The Hexam wetland saltmarsh systems in the Hunter River estuary supports significant fisheries for prawns and various finfish and crab species (Boys 2016, Taylor and Johnson 2016, Taylor *et al.* 2017). Recent work has shown that outwelled saltmarsh productivity supports a considerable proportion of fisheries productivity for the Hunter River (Raoult *et al.* 2018). Rehabilitation of the saltmarsh has demonstrated significant benefits from the recovering wetland to the commercial School Prawn fishery, with production from the site now comprising a substantial portion of the commercial harvest (Taylor *et al.* 2017).

Taylor and Creighton (2018) developed a coupled population-fishery model to explore the potential benefits to the School prawn (*Metapenaeus macleayi*) fishery of habitat rehabilitation in the Clarence River, NSW. Simulations showed restoring 27.6 ha of saltmarsh in this area could yield an annual recruitment subsidy which contributes up to 2,578 kg y^{-1} of additional School Prawn harvest, generating additional revenue of around $24,078 y^{-1} (GVP) and associated economic output of

$142,336 y^{-1}. These economic figures also do not include additional benefits from other ecosystem services. The restoration of connectivity to these areas will open up the habitat for direct usage by a broad cross-section of other commercially and recreationally targeted species, and these species will similarly benefit from any associated trophic subsidy.

Case Study 13: Saltmarsh restoration in San Francisco Bay

As part of regional efforts to reverse habitat loss, saltmarsh restoration was undertaken in South Bay, San Francisco to restore ecological functionality and productivity. Funds totalling $9.80 million were invested into restoration of 613 ha of former salt ponds. Monitoring has shown that restored ponds are beginning to provide vegetated marsh habitat and to support a different mix of bird, fish, and shellfish species, including threatened, endangered, and iconic species. Speers *et al.* (2015) estimated the value of changes in the commercial and recreational fisheries. Examining three key illustrative fish species, the annualised benefit to fisheries (3% discount, 40 years) was estimated to be $102,575 yr^{-1}. However, restoring the salt ponds to tidal wetlands will enhance a wide range of ecosystem goods and services for local and regional communities and provide total economic benefit of $3.29 to $12.34 million annually. The overall NPV for the project was estimated to be $81.42 million to $276.40 million, resulting in BCR of 9.3 to 29.1.

Cost benefit analysis using the estimated fisheries benefit ($1,095 ha^{-1} yr^{-1}) of saltmarshes from Janes *et al.* (2020) with the cost of saltmarsh rehabilitation ($83,465 ha^{-1}) from Bayraktarov *et al.* (2016), would generate NPV (30 years, 5% discount rate, 2 year delay for benefits to commence accruing) of $-68,668 ha^{-1}, and BCR of 0.17. This indicates that if saltmarsh rehabilitation was undertaken primarily for fishery enhancement it would likely not be economically viable. However, the benefits used do not include recreational value. If these were included it is likely economic feasibility would be achieved. A study incorporating both commercial and recreational fishery benefits from saltmarsh restoration is needed to confirm this assumption.

4.2.2.5 Coral reefs

Coral reefs provide economic benefits in terms of fishery output from directly and indirectly supporting exploitable finfish and invertebrate species (Spurgeon 2001). Coral reef rehabilitation is increasingly being used globally as a management tool to minimize accelerating coral reef degradation resulting from a range of local (pollution, sedimentation, overfishing) and global issues (*e.g.* climate change). The science of coral reef rehabilitation is still relatively young and very focused on ecological and technical considerations, with far less effort invested in socio-economic outcomes.

Despite plenty of literature highlighting the diversity and prevalence of exploitable species on coral reefs, specific economic values of fishery productivity per unit area has surprisingly, not been widely reported, especially for developed countries. The reported economic benefits to fisheries from coral reefs have been highly variable and ranged from $127 ha^{-1} yr^{-1} to $626,671 ha^{-1} yr^{-1} (Table 19). Many of the attempts to value fishery productivity from coral reefs have been undertaken in Pacific island or developing countries, and some of these estimated values have been extremely low. For example, the net value of reef fish was estimated to only be $226 ha^{-1} yr^{-1} in Sri Lanka (Berg *et al.* 1998) and $296 ha^{-1} yr^{-1} in Indonesia (Cesar 1996). Higher benefit values were reported from Jamaica ($6,776, Gustavson 1998), Philippines ($3,750 ha^{-1} yr^{-1}, White *et al.* 2009) and Vietnam ($5,665 ha^{-1} yr^{-1}, Ngoc 2019).The World Bank's ratios of purchasing power parity should be taken into consideration for more appropriate comparisons to account for the status of individual economies. This is especially the case when comparing values between developing and developed countries. Despite using this approach, Costanza *et al.*(1998) still reported a mean fisheries value for coral reefs of only $542 in their global review.

Other direct use values besides fishery harvest can be high for coral reefs. In a report into investment in coral reef preservation and enhancement, commercial fisheries generated 40% of the direct economic value associated with coral reefs in the Coral Triangle, but only 5% in the Mesoamerican Reef (UN Environment *et al.* 2018). Tourism generated the greatest economic benefits in both areas. These results reflect differences in the economic composition of countries in each region and highlight the potential ancillary benefits that could be generated from projects which rehabilitate coral reefs for fisheries purposes. In Montego Bay, Jamaica, the potential net revenue from commercial fishing was estimated at $7,394 ha^{-1} yr^{-1} (Gustavson 1998), whilst harvesting coral for the aquarium trade was worth around $1.4 million ha^{-1} yr^{-1} (Ruitenbeek and Cartier 1999) and recreational use estimated to be $1.7 million ha^{-1} yr^{-1} (Gustavson 1998).

More information was available on the costs of coral reef restoration. A simple comparison of the costs of past attempts at coral restoration reveals the potential magnitude and significant variation of costs involved (Table 19). Spurgeon and Lindhal (2009) found that costs for coral restoration schemes can vary from some $18,700 ha^{-1} to over $100's million ha^{-1}. Spurgeon (2001) argued that the economic efficiency of restoration can be enhanced by identifying and focusing rehabilitation efforts on those factors that affect the value of reefs which can readily be manipulated. For example, fishery benefits will generally be greater where reef structures provide additional voids and surface area for organisms to utilize and where they have higher live coral coverage.

A wide range of approaches have been utilised for coral reef restoration and the costs of using different techniques can vary greatly. Bortstom-Einarsson *et al.* (2018) found the median cost of coral restoration reported from 64 studies was $586,339 ha^{-1} ± 22,463,945 ha^{-1} (median ± SD, Table 23). The costs varied widely, ranging from as little as $5,518 ha^{-1} all the way to $177,806,200 ha^{-1}, depending upon the type of rehabilitation undertaken. Edwards *et al.* (1994) and Clark and Edwards (1995,1999) evaluated different options to rehabilitate sections of reef in the Maldives previously destroyed by coral mining. Rehabilitation was attempted by stabilising the substrate and installing artificial reefs and through coral transplants. Costs ranged from $1.0 million ha^{-1} for deployment of anchored chain link fencing, to $2.5 million ha^{-1} for concrete (Armourflex) mattresses, and up to $3.9 million ha^{-1} for the use of one cubic metre concrete blocks. Rehabilitating degraded coral reefs through transplantation of staghorn (*Acropora*) corals in Tanzania cost about $29,364 ha^{-1} (Lindahl 1998), excluding an initial one-time pre-construction cost for surveys, planning and training of the staff. The estimated costs were based on a hypothetical full-scale rehabilitation effort, which may not be required for fisheries enhancement.

In Australia, Kaly (1995) compared methods for enhancing coral cover using different coral transplantation techniques on tourist damaged coral reefs in the Great Barrier Reef. Increasing the natural density of corals on hard substrate by 10% was found to cost roughly $99,223 ha^{-1}. The estimated costs only included labour (diving) and materials used for the re-attachment. One diver could create 10% coral cover over 3.75 m2 per day. The costs did not include time for obtaining the corals or monitoring or damages to the donor sites.

Accurate reporting on the total costs of coral reef restoration has been inconsistent or omitted entirely within the published and grey literature reviewed (Table 19). The highest reported costs were typically for efforts to completely restore reefs to pristine condition following damage from vessel groundings. For example, NOAA (1997, 1999) reported restoration costs for repairing ship damage to be $14.3-260 million ha^{-1} (NOAA 1997,1999). These values may not be appropriate for fisheries enhancement objectives, where more affordable approaches may be able to generate the improved fisheries productivity that is sought.

Table 23 The costs of coral reef restoration associated with using different techniques. Adapted from Bostorm-Einarsson *et al.* (2018).

Restoration technique	**n**	**Restoration cost (2010 US$/ha)** Median (± SD)	Minimum	Maximum
Coral gardening	3	351,661 (± 136,601)	130,000	379,139
Coral gardening - Nursery phase	5	5,616 (± 22,124)	2,808	55,071
Coral gardening - Transplantation phase	2	761,864 (± 1,033,831)	30,835	1,492,893
Direct transplantation	21	761,864 (± 1,033,831)	4,438	3,680,396
Larval enhancement	6	523,308 (± 1,878,862)	6,262	4,333,826
Substrate addition - Artificial reef	15	3,911,240 (± 36,051,696)	14,076	143,000,000
Substrate stabilisation	8	467,652 (± 9,015,702)	91,052	26,100,000

Coral restoration studies have rarely included an economic benefit assessment, particularly with reference to changes in fisheries value (Table 22). No benefit cost analyses could be identified which examined the economic outcomes from a fisheries perspective. Most economic studies have instead focussed on the fishery impact from the loss of coral reef habitat at the macro-scale. For example, Alcala and Russ (1990) reported a decline of $151,389 in the total yield of reef fishes off Sumilon Island (Philippines) after breakdown of protective management. McAllister (1998) gives estimates of reef productivity for coral reefs in excellent condition (18 mt km^{-2} yr^{-1}) as well as good condition (13 mt km^{-2} yr^{-1}), and fair condition (8 mt km^{-2} yr^{-1}). Based on changes in condition over time and estimates of net profits associated with these yields, the estimated fisheries loss in the Philippines was $213 million yr^{-1}.

Global reviews of coral restoration literature have identified that one of the reasons benefit cost analyses are challenging to conduct is because reporting of costs and benefits is not standardized (Boström-Einarsson *et al.* 2018, Iacona *et al.* 2018). To date, reports on coral restoration success are mostly item-based (Bayraktarov *et al.* 2016), or reported in the form of cost effectiveness studies, which are methodologically different from benefit cost analyses (Abrina and Bennett 2021).

Several benefit cost analyses have been performed for coral reef rehabilitation without specific regard for the benefits to fisheries. Abrina and Bennett (2021) compared the net economic outcomes from two different rehabilitation techniques applied at either the local or national scale in the Philippines. The present benefits were based on the choice modelling method to estimate non-market (non-use) values. NPV for all rehabilitation approaches were positive, with BCR ranging from 5.3 to 20.9 using the conservative lower bound value estimates. This indicated that the coral restoration would be socio-economically feasible. Investment in mass larval enhancement were estimated to produce higher net benefits and BCR compared to those of coral gardening. In terms of scale, higher net social outcomes for the local-scale investments support more localized approaches to coral rehabilitation.

White *et al.* (2009) estimated the annual net revenue range from the Olango Island reef was \$593-982 ha^{-1}. Poor quality coral reef habitat on Olango Island was potentially limiting the economic value of the reef to the local community. Cost benefit analysis showed that investment in reef rehabilitation and management could significantly increase annual net revenues from fisheries and tourism expenditure. The estimated annual increase in management costs of \$141,000 yr^{-1} would be easily offset by the predicted doubling of the net revenue from commercial fisheries (extra \$583,000 yr^{-1}) alone within 5 years, and deliver a BCR of approximately 4.13.

De Groot *et al.* (2013) reviewed the data from 94 coral rehabilitation studies and estimated the total economic value derived from the direct benefits of coral reef ecosystem services was approximately \$500,000 ha^{-1} yr^{-1} (2% discount, over 20 years). Median restoration costs were estimated at approximately \$1.5 million, which produced a negative internal rate of return across most scenarios in a sensitivity analysis. Rehabilitation costs would need to be reduced by 21% to break even in most cases. This result suggests that coral reef rehabilitation for the sake of fisheries benefit would not be economically viable.

Cost benefit analysis using the estimated fisheries benefit (\$7,394 ha^{-1} y^{-1}) of coral reefs from Cesar (1998) with the mean cost of coral reef rehabilitation (\$586,000 ha^{-1}) reported by from Bostrom-Einarsson *et al.* (2018), would generate NPV (30 years, 5% discount rate, 10% progressive annual increase in benefit in first 10 years) of \$-500,317, and BCR of 0.15. These results confirm that coral rehabilitation is highly unlikely to be an economically viable fisheries enhancement tool for managers under most circumstances. However, the value of the benefits from coral rehabilitation to fisheries for Australia are unclear and no Australian examples of fishery value were identified. It is predicted that the value will be higher than the \$7,394 ha^{-1} y^{-1} used in the example above, and may produce similar economic analysis outcomes to that estimated above by de Groot *et al.* (2021).

The results from our review indicate that coral reef rehabilitation typically has high implementation costs, a long lag period before delivering benefits and has only been undertaken at a small spatial scale. There was very limited information economic impact available for past projects and no clear studies using benefit cost analysis could be identified. Non-fisheries benefits are likely to be significantly greater than those accruing to fisheries, and have been the primary driving pressure for rehabilitation projects. Development of production function for coral reef fisheries is recommended to inform future management decisions regarding the rehabilitation of coral reefs for the purpose of fisheries enhancement. These models would also assist to improve total economic value estimates for future coral rehabilitation projects.

4.2.2.6 Rivers

There is a substantial body of published literature describing the economic benefits of freshwater wetland rehabilitation, especially pertaining to river restoration. Much of the literature focusses on assessing non-use or non-market benefits and details pertaining specifically to fishery outcomes are less common. Most economic studies also do not separate out the contributions from different ecosystem service benefits in their analyses. Fisheries-based cost benefit analyses and economic impact assessments are therefore rare because of the relative scarcity of data relating to rehabilitation activities and changes in fishery value. The few studies available are somewhat difficult to compare because of large variations in the hydrological and ecological baseline conditions of rivers, socio-institutional settings, and methodological approaches, and they report mixed results, making it hard to conclude whether investments in river restoration projects can be economically justified from a fisheries enhancement perspective (Ayres *et al.* 2014, Logar *et al.* 2019).

One of the main reasons economic feasibility has rarely been reported is that river restoration can be expensive (Angelopoulos, *et al.* 2017, Table 19). Often multiple rehabilitation activities are required to restore degraded river reaches. Brooks and Lake (2007) reported the median costs for different types of river rehabilitation projects undertaken in Victoria ranged from $8,500 to $125,000 per project. The most expensive rehabilitation activities were fish passage, stormwater management and instream habitat improvement. More moderate costs were associated with water quality management, bank stability and channel reconfiguration, whilst education and riparian management cost the least. Lower cost activities, particularly riparian management, have been the most widely implemented. A cost analysis conducted by Washington Trout (2004) of 24 culvert rectification projects calculated the dollars spent for each mile of stream habitat opened ranged from $10,400 to $242,400 km^{-1}, with a mean of $64,395 ± 54,524 km^{-1}. Despite the rectification work being undertaken to support native fish populations, no fishery benefits were quantified in the analysis.

The scale of rehabilitation efforts also varies greatly, from hundreds of meters to thousands of kilometres of river, and can have a significant bearing of the response of exploitable fish species. Smaller projects may not occur at a large enough spatial scale to sufficiently address enough issues impacting on their populations. Ideally, rehabilitation activities need to be undertaken at the reach or sub-catchment scale to have a noticeable impact on fish populations, especially for more mobile species. Rehabilitation activities also need to address impacts at all life-history stages, to ensure successful spawning and recruitment to harvestable size.

River rehabilitation has been clearly demonstrated to improve the value of fisheries (Table 19). Hickley *et al.* (2004) detected noticeable improvement in fishery performance from the installation of floating reed rafts in freshwater urban waterways in Enfield Lock, London. Anglers considered that floating vegetation raft installation had increased their catch and improved their fishing experience. Competitors in a fishing competition caught significantly more fish from areas with rafts compared to anglers who didn't fish those areas and these results were supported by periodic electrofishing surveys.

Hoehn *et al.* (2003) highlighted that the economic value of freshwater ecosystems is derived from the services they provide, including fish production. Freshwater habitats are extremely valuable to many inland and diadromous commercial and recreation fisheries (*e.g.* Leahy *et al.* 2022). Acuna *et al.* (2013) found that at the reach scale the highest contribution to ecosystem service values in a river comes from opportunities for recreational fishing and the value of harvestable fish in a system. In the sites examined in the study, these fisheries values outweighed the ongoing maintenance costs to preserve and improve instream woody structure. Net benefits were also estimated to be larger in higher stream orders.

Van Vuuren and Roy (1993) estimated the fisheries value of freshwater marshland habitat in Ontario, Canada. Undyked marshes were identified as regionally significant for fish recruitment and growth, and supported the recreational angling in nearby Lake St. Clair. The NPV (4% discount over 50 year timeframe) of marshes was estimated to be \$13,466 ha^{-1} for hunting and fishing activities. Unfortunately the specific contribution from recreational fishing to this value was not provided.

The general population's willingness to pay (WTP) for improved river quality has been studied in many locations. In general there appears to be a strong community willingness-to-pay for river and freshwater habitat rehabilitation projects. In New South Wales, Bennett and Morrison (2001) found that WTP for increasing water quality from boatable to fishable was valued at a one-off payment of \$0.52 km^{-1} per in-catchment household and \$0.85 km^{-1} for out-of-catchment households (AUD 2000). Willingness to pay for increasing native fish numbers was \$3.55 per household. In the USA, numerous studies have reported on willingness to pay for fish passage restoration to improve the native salmonid fisheries. On the Elwha River, households were found willing to pay \$145-180 yr^{-1} for 10 years to improve salmon fisheries by an additional 300,000 salmon through the removal of two dams (Loomis1996). Hanemann *et al.* (1991) examined the WTP for a program designed to restore flows in the upper San Joaquin River, California, to enhance salmon, other fish and wildlife, and vegetation along the river banks. The salmon improvement project aimed to increase the number of Chinook salmon returning to spawn to 15,000 annually, and raise the total number of Chinook salmon (*O. tshawytscha*) caught by sport anglers (increase of 7,500 fish) and commercial fishers (increase of 23,000 fish). The truncated mean WTP was \$181 per year for California households. Similarly other studies have reported WTP values of \$7.32 to \$784.63 yr^{-1} per household to achieve substantial improvements in exploitable fish populations by improving fish passage for anadromous species (Tables 15 and 20). The retrospective mean WTP was \$241.63 ± 206.16 and the median WTP \$176.36 per household per year for increasing fish stocks in a river. Additionally, households had a mean WTP for river restoration of \$133.84 ± 107.44 per household per year. This compares favourably with the global mean of \$124.35 per household reported in the review by Brouwer and Sheremet (2017).

The mean BCR from the reviewed literature for fisheries benefit was strongly positive (6.34 ± 8.65), indicating high economic return on investment, but this was influenced by several outliers from small projects. The median value of BCR = 2.57 is probably more representative (Table 22). However, not all river habitat rehabilitation efforts for fisheries enhancement delivered positive ecological or economic net benefits. In some cases, the economic outcomes from river rehabilitation may not be positive unless the benefits realised from other ecosystem services are considered. Several studies focussing only on the recreational fishery benefits resulting from river rehabilitation have shown that the costs of restoration can far outweigh the benefits (*e.g.* Dubgaard *et al.* 2005, Paulrud and Laitila 2013).

A long-term population decline of Walleye (*Stizostedion vitreum vitreum*) in Lake Superior's Thunder Bay, Canada, was suggested to have occurred in part due to loss of spawning habitat (Schram *et al.* 1991). Habitat enhancement was undertaken to increase available spawning habitat. Surveys found that although Walleye eggs were deposited over a larger area, there was no evidence for augmentation of the total number of eggs deposited (Geiling *et al.* 1996). The habitat enhancement whilst utilised by the fish, had no significant impact on the Walleye population, resulting in a net negative economic impact for the project. Garner *et al.* (2013) suggested that restoring habitat connectivity would have been more cost effective in the long-term compared to stocking or enhancement of spawning habitat.

Dahlberg and Johansson (2006) analysed the net socioeconomic benefit of river restoration in Lilla Luleå River, Sweden, using cost-benefit analysis to determine if it would be socio-economically profitable to restore Lilla Luleå River for the purpose of recreational fishing. Rehabilitation involved improving river hydrology by releasing additional water from a hydro-electric dam and releasing fish to restore the wild population to its original condition. The cost-benefit analysis over a 20 year horizon (5% discount) showed that restoration was not socio-economically profitable based on increased recreational fishery benefit.

Despite negative results such as these, Bergstrom and Loomis (2017) found 68% of the river restoration projects they reviewed were conducted with fisheries enhancement as a primary objective.

No Australian cost benefit analysis studies on the fishery benefits from freshwater wetland rehabilitation could be identified. The few studies found, were undertaken in the USA and Europe and comprised of a mix of pre- and post-project analyses. The results have demonstrated that economic feasibility is possible from a fisheries enhancement perspective, under some circumstances.

In Idaho, the Big Wood River watershed restoration project proposal was developed to restore 40 miles of the river-channel to more naturally functioning conditions and enhance aquatic habitat. The objectives were to improve the recreational fishery, reduce flood risk and enhance other ecosystem services provided by the river. The project would involve channel reshaping and naturalisation, riparian revegetation and installation of large wood for bank stabilisation and fish habitat. Cook and Becker (2016) conducted an economic scoping analysis of the project, including the predicted marginal benefits for the recreational fishery. The cost of the project was predicted to be $22 million, and using the benefit-transfer method, the NPV of the economic benefit to anglers from the improved recreational fishery was estimated to be between $4.5 million and $32.4 million over 20 years (3-6% discount rate), giving BCR of between 1.21 and 1.81. The near-term economic impact to the local county from restoration associated construction was estimated to be $2.3 million in value added annually for 5 years, whilst long term economic impacts were estimated to grow to $1.9 million in value added annually over 15 years as the fishery improves. The analysis results suggest that the project would be economically feasible based on just the predicted benefits to the recreational fishery and would provide long-term economic benefits to the local county.

Holmes *et al.* (2004) assessed the benefits and costs of riparian restoration projects along the Little Tennessee River in western North Carolina. Mean rehabilitation costs for riparian buffer zone rehabilitation was estimated at \$50,500 km^{-1} for a combination of activities such as replanting of trees and grasses, installation of timber revetments, stock exclusion fencing and installation of offstream stock watering points. The WTP of local residents for 9.6 km of riparian rehabilitation was estimated using contingent valuation. The survey questions included valuations for the improvement of gamefish abundance, but specific values for this ecosystem service were not collected. The present value (5% discount, 10 year timeframe) for the aggregated median annual WTP for the rehabilitation activities was estimated at \$592,600 km^{-1}, generating an NPV of \$542,100 km^{-1} and BCR of 11.75.

Thomas and Blakemore (2007) conducted a cost benefit analysis for salmonid spawning habitat restoration in the Wye River, United Kingdom. The study compared the direct use benefits generated by recreational anglers, with the costs of restoration and lost farming output borne by farmers, using a contingent valuation method. Mean angler WTP was \$172.79 yr^{-1} and farmers did not perceive the habitat improvements as a significant threat to their agricultural production. The results of the cost benefit analysis gave a positive NPV indicating habitat rehabilitation would therefore be socio-economically profitable from a fisheries enhancement perspective for minor rehabilitation projects. However, the number of anglers using the Wye would need to double or triple in order to justify the full restoration budget that had been allocated in the Wye Habitat Improvement Project.

The socio-economic welfare effects from river rehabilitation to improve recreational fishing in three Swedish rivers was examined by Hnin (2017). A cost-benefit analysis was conducted using benefit transfer to estimate changes in angler welfare from previous WTP estimates. The accumulated total costs for habitat rehabilitation for fishery enhancement across the three rivers was estimated to be approximately \$9.2 million. Over a 10 year time horizon (3.5% discount rate), the NPV for the rehabilitation activities was \$14.5 million, with a BCR of 2.57, demonstrating positive socio-economic returns on the fisheries enhancement investment.

Brouwer and van Ek (2004) showed that the restoration of the floodplains of the lower River Rhine in the Netherlands was expensive and resulted in a BCR of 0.61 if only the financial implications are considered (discount rate of 4% over 50 years). The ratio becomes just higher than 1 (1.15) in a broader economic welfare analysis when consideration is also given to non-use welfare effects, such as biodiversity, habitat improvements and increasing the public's sense of safety.

Bellas and Kosnik (2019) conducted a retrospective benefit cost analysis on the benefits to native anadromous fish accruing from the Elwha River restoration project in Washington. The major components of the project were the removal of two large hydro-electric dams from the river system and the construction of a native fish hatchery and flood control structures. An original benefit cost analysis conducted prior to project commencement predicted that the harvest of all salmonid species from the river would increase annually, reaching a long-run equilibrium 20-25 years after dam decommissioning (Meyer and Lichtkoppler 1995). The net value of the increased fish harvest to the commercial, cultural and recreational sectors was predicted to be \$10.47 million, with most of the benefit accruing to the commercial fishers. The present value of those benefits was estimated to be \$32.0 million (7% discount, 100 year time horizon). The retrospective benefit cost analysis by Bellas and Kosnik (2019) determined significantly lower and delayed benefits and higher project costs to that from the original estimates. The present value of fisheries benefits was only \$9.45 million, whilst the project costs were \$427.0 million. Therefore, the likely fisheries NPV based on these more accurate values (7% discount, 100 year time horizon) would be \$-417.6 million, with a BCR of 0.02. Significant tourism and non-market benefits would need to be generated by the project for it to be economically feasible.

It is extremely difficult to identify a typical river rehabilitation scenario to economically analyse due to variation in spatial scale and the lack of representative costs and benefits associated with river habitat rehabilitation. Additionally, the unitised value reported in few studies conducted are per river kilometre, rather than per hectare. Instead, the median BCR of 2.57 from the literature review will be used for comparison, but without an associated NPV. The time horizons for river rehabilitation benefit cost analyses in the literature ranged from 10 years to 100 years, and thus assuming that the BCR would be relevant for a 30 year time horizon should hold.

4.3 Summary

This Chapter has demonstrated that strategically planned wetland habitat rehabilitation can make significant contributions to fishery productivity and deliver substantial socio-economic benefits under the right circumstances. However, habitat rehabilitation can be costly, and the benefits of restoration are rarely quantified sufficiently to understand whether these costs are justified. As highlighted by Tan *et al. (*2020), while sound polices and legislation may provide a firm foundation for upscaling habitat rehabilitation efforts, investment may be quickly undermined if resources are not carefully targeted to areas where threats to habitat persistence have been removed or reduced, successful habitat restoration is feasible, stakeholders are willing and able to invest, and the benefits to other environmental and social values are the greatest.

Given the often limited budget available for fishery enhancement objectives, a clear understanding of the costs and potential fishery benefits resulting from different habitat rehabilitation efforts is essential for resource managers to maximize investment returns. The aim of rehabilitation for fisheries should be to recreate key functional habitats and connectivity between these habitats, and the target should be a quality of environment that maximises productivity of fish (Cowx and Gerdeaux 2004). This is where the development of a coupled population-fishery models is needed to explore the potential fishery benefits from habitat rehabilitation using habitat–fishery linkages (Taylor *et al.* 2018, Taylor and Creighton 2019). The costs and feasibility of restoration projects over relevant spatial scales must be reliably estimated to ensure resources are invested optimally. Where possible, to achieve a significant net gain of wetland habitats and their associated fisheries values, the scale of rehabilitation should at least match the scale of degradation.

The high costs of complete restoration projects highlight the importance of considering an alternative, ‘partial restoration’ scheme. In partial restoration, the maximal ecosystem rehabilitation and public benefit are compromised for a more affordable combination of restoration elements (Becker *et al.* 2018). The question of whether to undertake complete restoration or compromise on partial restoration is a complex one, involving the assessment of interactions between its components (Wohl *et al.* 2015, McMillan and Noe 2017). This approach would be particularly relevant to habitat rehabilitation undertaken for fisheries enhancement. Activities could be targeted at delivering the maximum fisheries benefit for the minimum cost, rather than trying to develop the highest total economic value or outputs. Supplementary investment from other stakeholders with different output objectives could be used to deliver additional ecosystem service benefits.

Rehabilitation activities often needs to be prioritised between multiple habitats and geographical locations to maximise the benefits (Rogers *et al.* 2018). The benefits of rehabilitation are rarely quantified in consistent terms making it difficult to compare justification of costs between different projects or options. Standardised approaches are needed to make the investment value of restoration clear, potentially unlocking access to new financial resources for these activities. Although this Chapter has focussed on the provision of exploitable fishery resources, habitats also provide other valuable ecosystem services. Capturing the value of these and including them in cost benefit analyses may strengthen the case for undertaking habitat rehabilitation, especially where the fishery benefits alone may be insufficient economic justification. If a rehabilitation project is not effective in enhancing the overall level of ecosystem services, the derived economic benefits will probably be low (Holmes *et al.* 2004).

Creighton *et al.* (2017) outlined a framework for informing investment in habitat rehabilitation and restoration for fisheries. The framework shows how to make effective restoration decisions despite different levels of risk and uncertainty. The authors stress that whilst our biological understanding of the magnitude of stock increase associated with any specified repair action remains rudimentary, and predicting the return on investment is difficult, delaying decisions to invest also carries the costs of foregone benefits. Although the true magnitude of the fishery response may not be known, expert judgment can be used to estimate the probability of a discrete set of possibilities and estimate associated improvements in productivity or potential harvest. The results can be used to determine which investments are likely to deliver positive returns under different scenarios, and which are most likely to deliver the best return on investment or BCR.

Using cost benefit analysis will help establish an evidence-base to inform prioritisation of fisheries habitat rehabilitation Benefit cost ratios are used as the test metric in order to facilitate comparison across the differing investments. They can provide a guide to compare predicted results across the different restoration techniques. Table 24 summarises the indicative economic value to fisheries for rehabilitation of key habitat types based on cost benefit analysis outputs of typical scenarios applicable in Australia. The results show that mangrove rehabilitation is likely to generate the best return on investment from a fisheries perspective. Rehabilitation of rivers habitat, seagrass and shellfish reefs are also likely to provide positive economic returns for fisheries, but the high cost of coral reef restoration and the low value of fishery production from salt marshes mean rehabilitation of these habitats is not likely to be economically feasible for fisheries enhancement. These results are only indicative, and care needs to be taken because they are sensitive to the input values of the cost and benefits. Additionally, the exploitable species benefiting in each rehabilitated habitat type are not the same. Comparisons therefore should be based on the outcomes between habitat types for a similar species or species assemblage.

One other issue deserving consideration is that the benefit-cost ratio for preserving natural habitats can be as high or higher than rehabilitation activities (Su *et al.* 2021). Investing in maintaining and preserving existing natural habitats may in some instances be more cost-effective and deliver greater benefits than rehabilitating degraded systems. The costs and benefits from preservation should be included when conducting comparative cost benefit analyses, along with the base case of doing nothing.

The theory of diminishing marginal utility recognises that each additional unit of a commodity will be valued slightly lower than the unit before it – so the 1000th hectare of seagrass is worth slightly less than the 999th hectare, and probably much less than the 100th hectare (Hanley and Barbier 2009). Cost benefit analysis involves analysis of investments at the margin. Values estimated at the margin are likely to vary depending on the scale of the margin under investigation. For instance, as more and more area of habitat is restored, it is likely that the marginal benefit from an extra hectare of rehabilitated habitat will decline in line with the law of diminishing marginal utility (Abrina and Bennett 2021). Furthermore, as the scale of restoration investment increases, marginal costs may decrease as economies of scale are achieved. Cost benefit analyses conducted across differing scales must allow for such variations at the margin.

There appears to be considerable community support for utilising habitat rehabilitation to enhance fisheries. Non-market valuations have identified a willingness to pay amongst both users and non-users (Table 20). The aggregated value generated can be substantial and often significantly larger than the values derived from improved commercial harvests. Some of this value is likely attributable to the generally positive community sentiment towards environmental restoration, but some reflects the consumer surplus of recreational fishers who value their fisheries more than they are currently paying to access them. Such user or community support is vital to encourage backing by politicians and uptake of habitat rehabilitation as a fisheries enhancement tool by managers.

Table 24 The indicative economic value to fisheries for key habitat types. The net present values (NPV) were based on a standardised cost benefit analysis over a 30 year time horizon with a 5% discount rate. For saltmarsh, benefit accrual was delayed for 2 years whilst vegetation established. For seagrass the delay was only 1 year, whilst for mangroves, shellfish and coral reefs, the value of benefits accrued progressively increased annually by 10% for the first 10 years. No clear cost or benefit values were identifiable for river rehabilitation, so an indicative NPV could not be calculated. The BCR included was from the median of projects included in the review. All values have been converted to AUD 2021 for ease of comparison. The habitat values only refer to the benefits that could be achieved from habitat rehabilitation associated with commercial and recreational fisheries.

Habitat type	Habitat value ($ ha^{-1} yr^{-1})	Rehabilitation cost ($ ha^{-1})	NPV	BCR	Sources
Mangrove	20,092	48,469	184,359	4.8	Morton 1990, Bayraktarov *et al.* 2016
Seagrass	21,880	139,913	176,000	2.3	Janes *et al.* 2020a,b, Rogers *et al.* 2019
Shellfish reefs	27,533	223,933	95,122	1.42	Rogersi *et al.* 2018
Saltmarsh	1,095	83,465	-68,668	0.17	Janes *et al.* 2020, Bayraktarov *et al.* 2016
Coral reefs	7,394	586,399	-500,317	0.15	Cesar 1996, Bostrom-Einarsson *et al.* 2018
Rivers	n/a	n/a	n/a	2.57	Median value from this review

Chapter 5. The potential for fisheries enhancement in Australia

5.1 Need for fisheries enhancement

The over-riding goal of fisheries management is the long-term sustainable use of the fisheries resources to maximise socio-economic returns. Fisheries management is still commonly practised as a reactive activity, where decisions are made, and actions taken largely in response to problems that arise (Cochrane 2002). The resulting decisions are thus normally merely attempting to solve the immediate problems, with lower priority given to the broader perspective and the longer-term objectives. Incorporating fisheries enhancement strategies into management decision frameworks can help maintain and improve fisheries productivity, as well as address some of the other contemporary challenges facing aquatic ecosystems (Taylor *et al.* 2017).

Fisheries enhancement strategies expand the options available to fisheries managers beyond the use of traditional input-output controls. They provide opportunities for significant socio-economic benefits, through actively improving aquatic habitat and management of fish at the population level. Such approaches may simply offer alternative routes to a particular outcome (*e.g.* accelerating recovery of a stock that would also recover naturally), or they may support or create outcomes that cannot be achieved by other fisheries management measures (*e.g.* a high-value recreational fishery in a highly modified habitat). Enhancement strategies also have the potential to help manage the sometimes high social costs associated with harvest regulations (Beard *et al.* 2003, Johnston *et al.* 2011, Haglund *et al.* 2016).

This Chapter brings together the information on the three types of fisheries enhancement reviewed, summarising the relative costs, benefits and opportunities, both financial and non-financial. It compares quantitative socio-economic outcomes from each technique and discusses how they can be integrated to achieve synergistic results. The potential role of fishery enhancement in fisheries management is also examined. Significant interest and on-ground projects are already occurring in Australia regarding habitat enhancement, fish stocking and habitat rehabilitation. The Chapter concludes with a call for greater consideration for their strategic use as fisheries management tools, recognising that although we may never have the most precise scientific estimates of likely fisheries improvement flowing from many enhancement activities, we do know the benefits can be substantial.

5.2 Lack of socio-economic evaluation

Fisheries management is a complex socio-political process, and access to accurate, consistent data about how a fishery is performing, how a species is being managed, and the outcomes of those actions are a fundamental component for establishing effective management, regardless of the fishing sector or management system (Beddington *et al.* 2007). Quantitative assessment of enhancement contributions to fisheries management goals, such as increases in population abundance, yield, or economic rent, is essential. Accurate valuation allows the incorporation of sometimes otherwise unquantified values into decision-making frameworks. This can better inform decision-makers as to the full extent of the costs and benefits associated with proactive management of environmental resources, increasing the efficiency and effectiveness of decisions about their use to improve fisheries management and fisheries value. It is important to not only measure whether an enhancement strategy is meeting objectives, but also to assess the benefits relative to alternative management strategies.

One of the over-riding themes observed in this review was the consistent lack of quantitative socio-economic data reported on fisheries enhancement projects, both from Australia and globally. Key papers on fisheries enhancement all lament the lack of empirical data that has been collected, and highlight the importance of enhancement programs to quantitatively demonstrate cost-effectiveness. Such assessment is critical for effective fisheries management, because it permits the efficient allocation of management resources and enables comparisons between alternative management options. There still remain large knowledge gaps on the cost-effectiveness and economic outcomes for all three of the reviewed fisheries enhancement approaches. Outcomes from past projects appear to be heavily influenced by site-specific and scenario-specific factors, meaning that without sufficient case studies, transfer of benefits for appraisal of future projects entails high levels of risk and uncertainty.

Where economic appraisal or evaluation data have been reported, the approaches utilised and the costs and benefits included, have often been inconsistent. Most projects reported construction or purchase costs, but many did not include the inescapable costs associated with planning, administration and monitoring that are essential for making these projects to happen. Similarly, the range of benefits reported varied from fishery values for specific species through to the entire exploitable fish assemblage. Some studies also only provided benefits for fish greater in size than the minimum harvest threshold, whilst others included fish of all sizes. Non-use values were rarely examined or incorporated into analyses, primarily due to the cost and difficulty in collecting such data. Therefore, comprehensive economic analyses of fisheries enhancement projects were rare.

Whilst more action has been taken to enhance fisheries in freshwater environments, there has arguably been less scrutiny and evaluation of the outcomes of those efforts than in the marine environment. Quantitative assessments of habitat rehabilitation have rarely been undertaken from a fisheries stand-point. Better knowledge of the production function for the improved habitat is required to help justify the investment from fisheries managers. Community involvement in riverine rehabilitation is increasing (e.g. Ozfish Unlimited) and quantitative evaluation is important to better direct their efforts to achieve the greatest outcomes. In Australia, the majority of fish stocking occurs in inland rivers, but we still only have a rudimentary understanding of the impact this is having on the fishery. Apart from a few exceptions, we continue to struggle to understand basic information such as the contribution stocked fish make to recreational catches and what proportion of released fish survive. This lack of knowledge persists despite the significant ongoing investment from both governments and stakeholders into the production and release of additional fish.

For stocking of hatchery-reared fish, a primary limitation has been the ability to discriminate between wild and hatchery-reared individuals. There is still no national agreement on optimal marking methods for hatchery-reared fish. New developments in techniques which assess the genetic parental lineage of fish captured during surveys, hold great promise as a non-lethal monitoring tool (Fitzpatrick *et al.* 2023). The approach requires that the genetic profile of all broodstock be collected from hatcheries and maintained for comparison. Although possibly more costly than some physical or chemical marking techniques (e.g. calcein batch marking of otoliths, Crook *et al.* 2005, 2009), the approach has the benefits of being non-lethal, poses no risk of causing additional mortalities during the marking process, is easier for hatcheries, provides more information on the source locations of the fish, and can help monitor gene transfer through the receiving population (Fitzpatrick *et al.* 2023). It is recommended that efforts be made to adopt this approach nationally to overcome the lack of quantitative socio-economic data available, especially for stocking in non-impounded systems where outbreeding depression may have implications on the long-term fitness of a population. The exception to the use of genetic marking will be when wild eggs and larvae are captured and cultivated for re-release (e.g. Snapper *C. auratus* in Cockburn Sound, Western Australia). Alternative marking methods will need to be employed in such scenarios to ascertain the benefits achieved by the hatchery-rearing process.

5.3 Socio-economic outcomes of enhancement efforts

The three previous chapters have demonstrated that habitat enhancement, fish stocking and habitat rehabilitation all have the potential to improve fishery productivity and deliver positive socio-economic fishery outcomes. All three Chapters also included examples where using these enhancement strategies did not produce positive socio-economic outcomes based on fisheries investment and return alone.

Comparison of results from projects is difficult because the unitised value metrics applied varied between enhancement strategies. The base unit of artificial reefs differed between per reef, per hectare and per cubic meter of reef installed. In contrast, for habitat rehabilitation, values were typically reported per hectare, except for riverine rehabilitation which generally reported values per kilometre of rehabilitated river length. No standardised base units were apparent for reporting on socio-economic outcomes for fish stocking. These inconsistencies made comparison between the various economic analyses difficult, even within similar fisheries or enhancement approaches.

The one universal metric that could be utilised to compare across the different fisheries enhancement strategies was benefit cost ratio. This ratio is independent of project units or scale (excluding cost-efficiency savings that can occur in some larger projects). Overall, the BCR for fish stocking was almost twice as high as any other enhancement strategy (Table 25), indicating that investing in this strategy is likely to return the best economic value. This was followed by a habitat rehabilitation in mangroves, rivers, seagrass meadows and shellfish reefs. Installation of artificial reefs appears to be economically feasible, but likely cost neutral, whilst data for saltmarsh and coral reef rehabilitation indicate they are unlikely to be economically feasible from a fisheries enhancement perspective. However, there are a number of significant ecological risks associated with the release of hatchery-reared individuals and these have yet to be quantified and considered in benefit cost models. Valuing and incorporating such costs into economic analyses is likely to substantially reduce the economic feasibility of stocking, because restoration of such impacts is either impractical or extremely expensive.

The BCR for stocking recreational fisheries was by far the highest for any of the enhancement strategies examined. Given the relative expense of hatchery-based fishery release programs, it is very important that the benefits and costs associated with the activity are identified, quantified and effectively communicated (Taylor *et al.* 2017). As already highlighted, this has not happened sufficiently in Australia. Stocking targets specific species only and thus it is relatively straightforward to identify the end-users. The high BCR for stocking is likely due to the high value placed on recreational fishing and fish stocking by the recreational angler end-users (Garlock and Lorenzen 2017, WTP in Table 15), the direct fisheries link between fish stocking and consumer surplus (Gregg and Rolfe 2013), and creation of highly valuable new or improved fisheries in impoundments and lakes (e.g. Rutledge *et al.* 1990, Gregg and Rolfe 2013, Hunt *et al.* 2017). Garlock and Lorenzen (2016) found that in general, most anglers supported release programs, but the level of support for stocking compared with other fisheries management options, varied with the level of motivation and fishing intensity or specialisation of the angling group.

Stocking for commercial fisheries produced much more mixed results and the overall BCR, whilst still positive, was almost five times lower than that for recreational fisheries stocking. The necessary conditions for success appear to be far more restrictive and fishery specific. Internationally, large stocking programs for diadromous salmon species returned poor socio-economic results, whilst little difference in success was observed between stocking of finfish and invertebrates. Only limited commercial stocking currently occurs in Australia, but there are potential opportunities to generate positive economic returns on high-value species, particularly invertebrates such as lobster, abalone and scallops. Future research on improving survival of released individuals is necessary to make such stockings more broadly commercially viable.

In general, the return on investment for stocking was also closely linked with how connected the receiving system was, because this strongly influenced the recapture rate. Fully closed systems (impoundments and lakes) typically returned the highest benefits, followed by rivers, estuaries, bays and then open coastal and marine stocking. These results suggest that stocking should focus on enclosed or semi-enclosed waterways in order to generate the best economic return. In more open systems, emigration and the number of individuals that need to be released to generate a detectable benefit are limiting.

Strategically planned habitat rehabilitation can make significant contributions to fishery productivity and deliver substantial socio-economic benefits under the right circumstances, providing good justification for undertaking habitat repair for most habitat types (Table 25). However, habitat rehabilitation can be costly, and the benefits of restoration are rarely quantified sufficiently to understand whether these costs are justified from a fisheries perspective. Compared to stock enhancement and even habitat enhancement, habitat rehabilitation offers the potential advantages of a lower technical barrier to implementation and a wider range of benefits, including enhanced fisheries for a broader range of species and the additional ecosystem services provided by many habitats (Walters 2005, Walton *et al.* 2006a, Walton *et al.* 2006c, Barbier 2007).

Mangrove rehabilitation demonstrated the highest BCR, partially due to the high value of exploitable species frequently associated with that habitat. Coral reefs also have high-value species associated with them, but the costs for coral reef rehabilitation are so high as to make it economically unfeasible from a fisheries enhancement perspective. Rehabilitation efforts for rivers and seagrass meadows overall provided good returns on investment, whilst shellfish reef restoration provided more moderate socio-economic returns to fisheries, despite delivering substantial productivity increases. Again, the cost of reef construction limits net economic value. Only limited information was available on the economics of saltmarsh rehabilitation, but it suggested that the fisheries benefits were clearly outweighed by the costs in most cases.

Unfortunately, the scale of rehabilitation has often been at an experimental level (<1ha), meaning that substantial contributions to regional fisheries were rarely detected. Investment for upscaling may be quickly undermined if resources are not carefully targeted to areas where threats to habitat persistence have been removed or reduced, successful habitat restoration is feasible, stakeholders are willing and able to invest, and the benefits to other environmental and social values are the greatest.

Globally, habitat enhancement is generally considered an economic asset by stakeholders and managers, who are usually willing to contribute towards construction and maintenance. The often high construction cost and attraction to non-resident anglers has typically led to high economic outputs for regions where structures had been installed. The potential of reef complexes to attract exploitable species had socio-economic benefits for artisanal and recreational fishermen, but few examples of economic feasibility could be found for larger scale commercial fishing reefs, outside of ranching projects. In Australia there is an increasing trend to install more habitat enhancement structures due to demand from the recreational sector. However, only limited socio-economic evaluation has been conducted on Australian habitat enhancement projects so far. Anglers have a high willingness to fund creation of new habitat enhancement sites, using both purpose-built reefs and FADs, but the recreational fishing value for artificial reefs was often lower than that of natural reefs (Table 6). This suggests that in some instances, better value may be obtained by protecting and rehabilitating natural reefs, rather than creating new complexes. The costs for installing artificial reefs in marine areas is typically quite high, with most reefs in Australia costing over $1 million to complete. FADs cost far less to install, but their socio-economic impacts on recreational fisheries have yet to be evaluated. The median benefit cost ratio for artificial reefs whilst positive, generally remains low. Median BCR values across all studies was only 1.29, whilst recreational fishing reefs in Australia were essentially cost neutral, with a BCR closer to 1.

Analysis of artificial reef projects sometimes included economic impact assessment to investigate the benefits realised to the local economy. This form of economic analysis was rare amongst the other fisheries enhancement strategies, possibly because the links of the other enhancement activities to local economies are harder to quantify. The economic impacts from installing artificial reefs and FADs were generally positive, delivering significant jobs, income, expenditure and value added (Table 4), particularly for recreational fisheries. The size of the economic impact was closely related to the scale of the habitat enhancement activity, but there was also a trend for declining marginal benefits with scale (Sutton and Bushnell 2007).

In summary, enhancement activities which target recreational fisheries are more likely to return higher benefit cost ratios than comparable activities for commercial fisheries. Fish stocking resulted in the most favourable socio-economic returns on investment, but contained the greatest ecological risks. Rehabilitation of mangroves, river habitat, seagrass meadows and shellfish reefs should return positive economic outcomes for fisheries, but saltmarsh and coral reef rehabilitation are unlikely to be economically viable. Artificial reefs, whilst extremely popular with recreational anglers, currently have limited value for most commercial fisheries, and are likely to generate only slightly positive socio-economic outcomes unless they increase fishery productivity substantially. Artificial reefs can also pose an ecological risk from overharvest if appropriate management policies are not also put in place.

Table 25 Summary of the indicative economic values for each enhancement strategy from the standardised costs benefit analyses. NPV were calculated at 5% social discount over a 30 year time horizon, except for offshore artificial reefs which used a 7% discount rate. All values are in AUD 2021.

Enhancement strategy	Units	Habitat value	Enhancement cost	NPV	BCR	Comments
Offshore artificial reefs	Per reef (1,600 m^3)	55,900,000 use value over 30 years	1,100,000	200,461	1.01	BCR = 1.1-1.18 on construction costs alone
Fish stocking – all fisheries	Not standardised per unit				7.41	
Fish stocking – recreational	Not standardised per unit				11.83	
Fish stocking - commercial	Not standardised per unit				2.43	
Mangrove rehabilitation	Per hectare	20,092	48,469	184,359	4.8	
Seagrass rehabilitation	Per hectare	21,880	139,913	176,000	2.3	
Shellfish reef rehabilitation	Per hectare	27,533	223,933	95,122	1.42	
Saltmarsh rehabilitation	Per hectare	1,095	83,465	-68,668	0.17	
Coral reef rehabilitation	Per hectare	7,394	586,399	-500,317	0.15	
River rehabilitation	Per kilometre	n/a	n/a	n/a	2.57	Unitised habitat value and costs not available

5.4 Importance of understanding and addressing underlying issues affecting fisheries productivity

The review has highlighted how a comprehensive bio-economic understanding of a fisheries system is important for optimising outcomes from enhancement strategies. Identifying the underlying causes for underperforming fisheries or matters which restrict fishery expansion, is essential if actions are to be undertaken efficiently and cost-effectively. Knowing if recruitment, predation, resource limitation or spawning constraints are limiting fisheries productivity informs selection of the potential suitable enhancement strategies to apply to achieve the management objectives. A clear understanding of issues also enables the anticipated outcomes associated with different management scenarios to be compared. Enhancement may not always be the most cost-effective approach and needs to be compared with the results achievable through traditional management and other actions that could address specific issues. In some scenarios, none of the fisheries enhancement options may be viable options because they do not address the underlying cause of the issue or would be too costly to implement. Liu *et al.* (2019) found that in some inland and coastal waterways, strict pollution control might yield the largest benefits for fisheries and contribute the most to the regeneration of degraded aquatic ecosystems and fisheries. The second most promising scenario was the introduction of a no fishing season and habitat conservation. Fish stocking to bolster stocks was only the third best option.

Stocking effectiveness is critically dependent upon habitat availability and quality. A common theme amongst published reviews has been that provision of sufficient suitable habitat should be the number one priority for enhancement activities. Where fish numbers have been reduced (*e.g.* through overfishing or environmental degradation), addressing the primary causes for stock decline needs to occur prior to the release of additional individuals into the system. If juvenile recruitment limitations are a primary mechanism in establishing fish abundance in an area, then hatchery releases should increase abundance. Conversely, if habitat is a primary factor limiting the abundance of certain habitat-associated fish populations, then hatchery releases will be ineffectual, and efforts would be better invested in fixing the habitat issues. There is no point in stocking if there is insufficient habitat (quantity and quality) available to support the additional fish. Stocking is unlikely to deliver significant benefits unless the habitat limitations are addressed, or fish can be stocked at a size which bypasses the reliance on the limiting habitat.

Considering the spatial scale and nature of the fishery being targeted is also important when developing management objectives. Stock enhancement operates at the species level, whilst habitat enhancement and rehabilitation impact at the ecosystem scale and have the potential to benefit multiple exploited species and the ecosystem services that support them. Stock enhancement generally also has higher ongoing costs, whereas most of the cost for habitat projects is incurred up-front. Therefore, the addition or rehabilitation of habitats may return higher long-term fisheries value because they influence more exploitable species over a greater length of time. Additional benefits are also likely to accrue in habitat projects through improvements to the ecosystem services delivered besides those directly supporting fisheries production.

5.5 Integration of approaches

Lorenzen *et al.* (2013) proposed that controlling fishing effort, habitat management (restoration, rehabilitation and deployment of artificial habitats), and fish stocking are the three principal means by which fisheries can be sustained and improved. However, it is possible that multiplicative gains may be made through a combination of these approaches. Fishery enhancement approaches should not be seen as a replacement for good fishery management, but instead as part of a suite of potential management tools that can be utilised together to deliver strong, sustainable fisheries outcomes. Different fisheries enhancement strategies do not need to be undertaken in isolation and seen as alternatives to each other. The greatest benefits are likely to occur when different strategies are integrated to comprehensively address the issues limiting fisheries production or expansion. Depending on the situation, employment of different enhancement approaches or combinations of approaches may improve the likelihood that enhancement or restoration goals will be achieved. It is therefore desirable to move toward developing a common framework for integrating enhancement approaches.

Evidence of this potential may lie in some recent examples where artificial habitats have been deployed specifically to enhance the outcomes from hatchery-releases. For example, the fisheries and socio-economic benefits derived from habitat rehabilitation provide a good justification for undertaking habitat repair, but the full realisation of potential fisheries benefits relies on an adequate supply of wild recruits to utilise that habitat (Taylor *et al.* 2017). Inadequate supply of wild recruitment could be overcome by also combining stock enhancement with the habitat provision. The release of exploitable species that do not readily recruit to rehabilitated sites, but naturally occur in such habitats, may help to enhance the benefits of the habitat rehabilitation. The successful stocking of native fish in impoundments where they cannot naturally recruit is a great example of this.

A more commercial approach to this is ranching, where artificial habitat is introduced with the intent of supporting fish stocked for both commercial and recreational harvest. As in the above example, inadequate wild recruitment may limit the magnitude of fisheries enhancement that can be achieved from the introduction of habitat alone. Stocking can bypass this recruitment limitation to deliver better yields. While stock enhancement represents the effort to improve annual recruitment, ranching represents an attempt to increase the annual yield of a species (Moskness 1999). Ranching is similar to farming, in that stock is released into areas of improved habitat (or pasture) to grow and then be harvested. The Ocean Grown Abalone ranching project in Western Australia, provides a great commercial example of the economic benefits that can be achieved by combining habitat enhancement and stock enhancement in a suitable scenario (Case Study 14).

Large-scale commercial ranching has been widely used in China, Japan and South Korea (Seaman *et al.* 2011, Kim *et al.* 2017, Kitada 2020), and historically focussed mainly on non-migratory invertebrate species. Greater attention is now being given to incorporate non-migratory finfish into programs (e.g. Kim *et al.* 2017). As described in the Chapter 2 on habitat enhancement (section 2.3.2.1), establishing marine fishing ranches can deliver significant socio-economic benefits from the resulting recreational and commercial fisheries. Ranching is considered to be at the boundary between capture fisheries and aquaculture. This boundary is becoming less distinct as natural habitats are modified by the introduction of artificial reefs (Bartley and Bell 2008).

In Korea, using the combination of artificial reefs and stock enhancement in the Gyeong-Nam Province has delivered significant economic impacts. Construction of the Large Sea Ranch increased commercial fish production more than five-fold and increased fishermen's incomes by 26% (Kim *et al.* 2017). Additionally, recreational anglers spent $26 million at the site (Pyo 2009).

Although rarely used in Australia, this integrated approach to fisheries enhancement has received growing attention, especially following the success of commercial abalone ranching in the south-west of Western Australia (see Case study 14).

Case Study 14: Combining habitat enhancement and stock enhancement in Greenlip abalone

Out-planting abalone seed on artificial reef can potentially achieve the dual aim of stock enhancement and habitat improvement. A good contemporary example is the approach taken to enhance Greenlip Abalone (*Haliotis laeviagata*) production in south-western Australia by Ocean Grown Abalone (Hart *et al.* 2015, Hart and Strain 2016). This operation successfully combines commercial leases of seafloor area with the development of patented purpose-built artificial reefs and hatchery-based stock enhancement. Initially 5,000 reefs modules were deployed, but initial production has been so successful, producing more than 100 t of abalone, that expansion has already been developed. Trials were also undertaken in South Australia, but were deemed unsuccessful due low survival and slow growth, relative to equivalent trials in Western Australia (Burnell *et al.* 2019). High levels of predation and low food availability were considered the primary causes of poor survival and growth. This highlights the need for comprehensive understanding of the receiving environment for the application of fisheries enhancement to occur successfully.

5.6 Improving cost efficiency

Undertaking fishery enhancement projects can be expensive and reducing implementation and ongoing costs will improve long-term economic viability and overall socio-economic outcomes. Unfortunately, the lack of quantitative information on fisheries enhancement projects somewhat limits our ability to identify potential ways to conduct projects more cost efficiently. Further research and socio-economic evaluation need to be included in future projects to enable more informed management decisions and assist in selection of the most cost-efficient enhancement approaches to achieve management objectives in a given situation.

Stakeholders have often shown interest in and strong support for fisheries enhancement projects, and community groups are often willing to volunteer their assistance to make projects happen. Engaging with stakeholders throughout the planning process will encourage such participation and can greatly reduce labour costs and increase cost-efficiency. Although volunteers are mostly considered to assist with rehabilitation projects, they commonly also play a major role in fish stocking (e.g. fish stocking clubs in Queensland) and even habitat enhancement (e.g. fishing clubs building and maintaining fish attractors for impoundments or offshore).

Opportunities also exist to improve the cost-effectiveness of fish stocking programs. Cost efficiency gains can be achieved primarily through improved survival and fitness of released fish and reducing production costs. Acclimatization and pre-release conditioning programs can improve the survival, growth and reproductive success of hatchery-reared fish, providing a better return on investment. Identifying and stocking fish and invertebrates at the optimal size and time can also improve cost efficiency. For a few species, collection of highly abundant wild larvae and post larvae can provide a more cost efficient way to produce stock for release than spawning from broodstock, and at the same time reduce the risk of genetic impacts on the wild population.

Integration of enhancement strategies has the potential to increase the cost efficiency of management efforts. Combining habitat enhancement with stocking can overcome natural limitations in both recruitment and habitat availability, delivering multiplicative gains. In areas with poor or degraded habitat, habitat enhancement or rehabilitation prior to the release of the hatchery-reared juveniles will improve survival, growth and carrying capacity. This is especially important for the nursery areas utilised immediately upon release.

5.7 Need for appropriate management to realize maximum benefit

One of the risks of improving fisheries productivity or harvest efficiency in an area is the potential for increased fishing pressure. In open access fisheries, the distribution of fishing effort is likely to dynamically shift in response to increased resource availability. Sites which have received habitat enhancement activities have high potential to experience increased harvest pressure and overexploitation unless appropriate management practices are put in place (Whitmarsh and Pickering 2000, Lorenzen *et al.* 2013). This additional effort may offset gains achieved from the additional habitat, especially if aggregation of key species is not supported by corresponding increases in local productivity and recruitment. Milon (1989) describes this as the ‘paradox of artificial reef development’, where producing reefs that are biologically effective may jeopardise the overall economic performance of a fishery if access to the resource is not controlled. This is particularly the case where regional fish stocks are either fully exploited or overexploited. Locating habitat enhancement structures in protected areas or restricting access may prove more socio-economically beneficial than permitting open access (Taylor *et al.* 2017).

A great example of incorporating management objectives into habitat enhancement design has occurred at the Algarve artificial reef complex off Portugal. The artificial reef there has been constructed and managed as two separate zones with different designated purposes (Santos and Monteiro 1997, Whitmarsh *et al.* 2008). One section of the reef is designated as a no-fishing conservation zone (protection reef) and contains a high density of smaller artificial reef modules in shallower water to promote productivity. Adjacent to this is a broader field of more scattered larger reef structures which is designated as the exploitation zone. The objective of the protection reef is to generate the fisheries productivity necessary to support the harvest occurring in the exploitation reef system. The key behind making this management approach work is ensuring that the fisheries productivity of the protection reef is of sufficient to offset the increased harvest pressure in the exploitation reef. This is likely to require the protection reef to be larger in size than the exploitation reef. Pitcher *et al.* (2002) found that while small, protected areas with artificial reefs achieved little to avert collapse of the fisheries or a shift towards catches of low-value species, larger protected areas can do much to restore valuable fisheries for reef-associated fish.

There is potential to use the combination of protection and exploitation reefs in Australia to produce more productive fisheries. The initial construction costs will be significantly higher, but the longer term benefits from higher yields and sustainability may provide better long-term socio-economic outcomes.

5.8 Development of decision support frameworks and adaptive management

Different fisheries stakeholders can be strongly divided in their perception of the utility of fishery enhancement strategies (Hilborn 1999, Leber 2002, Arlinghaus and Mehner 2005, Lorenzen 2005, Hasler *et al.* 2011). Decision support tools can help fisheries managers identify the best management options in a transparent and justifiable way. Decision-support frameworks and modelling tools already exist which can be readily adapted to assess the outcomes of fisheries enhancement scenarios (*e.g.* Lorenzen 2008), and evaluate the ability of hatchery releases to integrate and add-value to habitat rehabilitation or enhancement. Application of these tools to evaluate the potential for multiplicative gains and the relative costs and benefits of such endeavours, will allow informed decision making prior to any large investments being made.

A survey of fisheries managers across Australia indicated that there was a strong desire for access to specific decision support tools to better incorporate fisheries enhancement options into current fisheries management decision frameworks (Norris 2023). However, there was no consensus on the level of complexity of such decision support tools that would be universally appropriate. The level of detail required by managers, ranged from simple decision matrices to help identify which enhancement approaches were appropriate to a particular fishery, through to highly detailed bio-economic models whose parameters could be manipulated to estimate the fishery and socio-economic outcomes across various management options and response scenarios. Disagreement also existed on whether such tools should be stand-alone or able to fully integrate into existing frameworks.

Bio-economic modelling is key to appraising the potential of habitat or stocking initiatives relative to other fisheries management measures and evaluating the cost–benefits of individual programs (Broadley *et al.* 2017). Several excellent contemporary studies in Australia have used bio-economic modelling to appraise the viability of proposed fishery enhancement projects (e.g. Ye *et al.* 2005, Hart *et al.*2013, Broadley *et al.* 2017, Taylor *et al.* 2018, Raoult *et al.* 2022). Management actions such as harvest regulations and stocking will likely not be equally effective at achieving various objectives, because they differentially affect fish populations and fishers (Lorenzen 2014).

Whilst this review and many others focus on cost benefit analysis to compare socio-economic outcomes between fisheries enhancement options, it may not always be the most appropriate approach. Where there are clear and commonly agreed objectives or targets to be reached in a specific policy or management context, then the most appropriate approach is likely to be cost-effectiveness analysis (Ledoux and Turner 2002). On the other hand, when targets cannot be pre-defined but must be determined within the assessment exercise, and all or most of the impacts can be expressed in money terms then cost–benefit analysis will be favoured. A comprehensive adaptive management framework is required to guide fishery enhancement decisions about the most appropriate tools and approaches to use, integrate priorities and flexible respond to new scientific knowledge arising from monitoring programs (Lorenzen *et al.* 2010, Talley *et al.* 2018).

Given the generally species-specific and site-specific outcomes from many enhancement activities, a two-step process may be merited to improve cost efficiency. An initial support tool can be used to rapidly identify potentially suitable enhancement strategies, based on the threats, issues and limitation of the specific fishery in question. Fishery-specific bio-economic modelling can then be used to examine how the suitable enhancement strategies can best be integrated using cost-benefit analysis or cost-effectiveness analysis.

5.9 Opportunities in Australia

A measured and responsible approach to the employment of fisheries enhancement strategies has generally been undertaken in Australia, particularly in the past two decades. Comprehensive research and planning now underpins most new enhancement projects (e.g. NSW marine stocking strategy, WA artificial reef program etc.). However, better data collection on the socio-economic impacts is required to facilitate greater uptake by fishery managers and develop support from stakeholders. New projects need to incorporate socio-economic appraisal or evaluation as a core component of their design, not only to further our knowledge base, but to also justify to stakeholders and investors that the expenses outlaid have been warranted and will provide a positive socio-economic return.

Clearly understanding the threats and stressors impacting fisheries systems, identifying realistic and quantitative management objectives and increased use of bio-economic modelling will be core to pro-actively managing commercial and recreational fisheries in Australia in a sustainable way using fishery enhancement strategies. Incorporating enhancement options into decision making processes will expand the options available to fisheries managers beyond the use of traditional input-output controls.

There is potential to expand the value of recreational fisheries and create niche fishing opportunities that can drive regional development. Rehabilitation of aquatic habitat has the potential to sustainably increase the productivity, yield and value of Australia's fisheries and improve resilience against adverse events and climate change. Fish stocking has significant potential for expansion, particularly in estuarine and impoundment systems. There are opportunities to expand the suite of species that are currently stocked, to diversify recreational fishing opportunities and attract more angler to areas with unique fisheries. Artificial reefs and FADs currently have high recreational fisher support, and their installation can improve access and fishing options. Undertaken appropriately, habitat enhancement can also increase overall fishery productivity by supporting the life-history requirements of fish and invertebrates.

For commercial fisheries, habitat rehabilitation has the potential to increase wild recruitment that may be limited due to the degradation and loss of essential fish habitats, such as nursery areas. Better recruitment is likely to result in increased yields and greater long-term sustainability within fisheries. Stock enhancement has the potential in some species to help recover depleted wild stocks more rapidly or to create new fisheries. The greatest potential for stock enhancement in the short-term remains with stocking less mobile, high-value invertebrate species.

Fishery enhancement approaches should not be seen as a replacement for good fishery management, but instead as part of a suite of potential management tools that can be utilised together to deliver strong, sustainable fisheries outcomes. Integrating different fishery enhancement strategies has the potential to deliver substantial socio-economic benefits. The greatest benefits are likely to occur when different strategies are integrated to comprehensively address the issues limiting fisheries production or expansion. Combining management of habitat to increase carrying capacity and responsible stock enhancement to overcome recruitment limitations will help optimise stock levels and harvest potential in the most efficient way.

In this review enhancement activities have been viewed purely through the lens of the resultant socio-economic benefits to fisheries. However, enhancement activities can deliver a broad range of ecosystem service benefits which provide substantial value beyond fisheries. These activities can also improve environmental health, species conservation, and support provision of a suite of other environmental, social, recreational and commercial opportunities. The cost benefit results in this review have deliberately not taken these additional benefits into account, in order to provide a clearer picture of the outcomes for fisheries. However, the substantial additional or value added benefits that can be generated can be used to seek and justify co-investment in projects from non-fishery sectors. Collaborating with relevant non-fishery stakeholders has the potential to greatly reduce the direct contribution costs of fisheries managers for some enhancement projects, which would lead to significantly better benefit cost ratios than those reported in this review. Where possible fisheries enhancement projects should pro-actively seek support from non-fishery sectors to more cost-efficiently achieve their fishery management objectives.

Ultimately, the success of fisheries enhancement projects will “reflect the quality of the prior planning and ongoing management” (Baine 2001).

5.10 Recommendations

This review provides the justification for using fisheries enhancement strategies in Australia, based on the socio-economic benefits they can deliver. However, further work is recommended to deliver greater adoption by fisheries managers and to cement consideration of their use into management decision frameworks. Further research is needed to understand where and how they can be most cost-effectively applied. The following recommendation will help clarify our understanding on the outcomes of fisheries enhancement activities and enable more cost-effective implementation:

- The lack of socio-economic data on fisheries enhancement options is currently limiting our understanding and their use. All new major projects should incorporate some form of socio-economic analysis to understand the outcomes of the activities, and develop a knowledge database that can assist in the feasibility analysis of new projects.

- To aid better comparison of the socio-economic outcomes from enhancement activities, standardised analysis and reporting guidelines should be developed to provide consistent and comparable results.

- Large-scale fishery enhancement is still not an exact science, with many potential variables which can confound predicted results. It is recommended that enhancement projects employ an adaptive management framework to enable the results from monitoring and research to be rapidly adopted. This will help projects to deliver optimal outcomes, especially given the levels of risk, uncertainty and environmental variability that occurs in aquatic ecosystems.

- A suitability matrix for fisheries enhancement options should be developed for all fisheries in Australia to provide managers with a rapid method for identifying appropriate enhancement strategies. The concept proposed by Grant *et al.* (2017) for mapping constraints across the various life-history of a target species, can be used to identify and prioritise issues where enhancement activities will have the greatest benefits. Similarly, the process used by Florisson *et al.* (2018) to evaluate the potential benefits of man-made structures for different Western Australian fisheries could be modified to include fish stocking and habitat rehabilitation options, and applied to all fisheries at the national scale.

- Investigate the potential of developing a generic bio-economic model and an associated database containing relevant biological and economic parameters for a range of species and fisheries. This may facilitate easier and faster comparison of potential enhancement options by fisheries managers.

- Co-investment for fisheries enhancement projects should be sought from various stakeholder groups where relevant, to capitalise on the broad ecosystem services that can be delivered.

Rehabilitation projects can deliver significant conservation benefits, whilst habitat enhancement projects have the potential for significant tourism value through diving and other viewing activities. They provide great opportunities for joint-funding.

- The use of different fisheries enhancement strategies should be integrated to potentially deliver multiplicative benefits across the entire life-history of target species. Habitat and recruitment are both essential to achieve sustained fishery outcomes, and enhancement projects should integrate both where possible to improve fitness across the entire life-cycle of target species.
- A national approach to marking hatchery-reared fish is critically needed. Despite millions of fish being released annually, the ability to track the outcomes from stocking remains limited. The origin of all fish released must be able to be traced. Genetic marking through lineage and parental analysis holds great promise and should be considered for adoption as a national approach (e.g. Fitzpatrick *et al.* 2023).
- Implementation of stocking cost-efficiency gains should be undertaken by improving release strategies through better survival and fitness from pre-release training, acclimation and improvement of release habitats.

Table 26 Summary of the indicative economic values for each enhancement strategy from the standardised costs benefit analyses. NPV were calculated at 5% social discount over a 30 year time horizon, except for offshore artificial reefs which used a 7% discount rate. All values are in AUD 2021.

Enhancement strategy	Units	Habitat value	Enhancement cost	NPV	BCR	Comments
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Seagrass rehabilitation	Per hectare	21,880	139,913	176,000	2.3	
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Saltmarsh rehabilitation	Per hectare	1,095	83,465	-68,668	0.17	
Coral reef rehabilitation	Per hectare	7,394	586,399	-500,317	0.15	
River rehabilitation	Per kilometre	n/a	n/a	n/a	2.57	Habitat value and costs not available

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Appendix A – Contribution of stocking to fisheries

Table 27 Impact on and contribution of stocking to fisheries (± standard deviation) for past enhancement projects.

Species	Location	Environment	Fishery type	Contribution to fishery (%)	Change in fishery	Source
Abalone spp. *Haliotis spp.*	Japan	Coastal	Com	29.12 ± 14.65% (6.9-83.8%)	n/a	Hamasaki & Kitada 2008
Abalone spp. *Haliotis spp.*	Japan	Coastal	Com	28.4 ± 10.4%	n/a	Kitada 2020
Barramundi *Lates calcarifer*	Johnstone River, Australia	River	Com Rec	10% 15%	n/a	Rimmer & Russell 1998
Barramundi *Lates calcarifer*	Dry Tropics, Australia	Impoundment stocking, but estuary fishery	Com	3%	n/a	Leahy *et al.* 2022
Black bream *Acanthopagrus butcheri*	Blackwood River, Australia	Estuary	Com Rec	32–74%	n/a	Cottingham *et al.* 2015
Black crappie *Pomoxis nigromaculatus*	Tennessee, USA	Reservoir	Rec	55% (0-93%)	n/a	Isermann *et al.* 2002
Black rockfish *Sebastes schlegeli*	Yamada Bay, Japan	Coastal Bay	Com	38.3% (4.9-73.5%)	Increased	Nakagawa *et al.* 2004
Brown trout *Salmo trutta*	Tasmania, Australia	River	Rec	1-2%	n/a	Douglas & Lieschke 2016
Chinook salmon *Oncorhynchus tshawytscha*	Lake Superior, USA	Reservoir	Com, Rec	25%	n/a	Peck *et al.* 1999

Species	Location	Environment	Fishery type	Contribution to fishery (%)	Change in fishery	Source
Chinook salmon *Oncorhynchus tshawytscha*	New York, USA	River	Com, Rec	68%	n/a	Nack *et al.* 2011
Common snook *Centropomus undecimalis*	Estuary	Rec	Rec	41.2 ± 8.6% 15.20 ± 6.2%	2 x increase 1.1 x increase	Brennan *et al.* 2008
Dusky flathead *Platycephalus fuscus*	Maroochy River, Queensland	Estuary	Com Rec	28% 47%	Inconclusive due to fish kills	Butcher *et al.* 2000
European lobsters *Homarus gammarus*	Norway	Open coastal	Com	50-60% local markets	1.7 x increase	Agnalt *et al.* 1999, 2004
Golden perch *Macquaria ambigua*	Murray River, NSW	River	Rec	9%	n/a	Forbes *et al.* 2016
Golden perch *Macquaria ambigua*	Murrumbidgee River, NSW	River	Rec	14%	n/a	Forbes *et al.* 2016
Golden perch *Macquaria ambigua*	Burrinjuck Dam, NSW	Reservoir	Rec	23%	n/a	Forbes *et al.* 2016
Golden perch *Macquaria ambigua*	Copeton Dam, NSW	Reservoir	Rec	98%	n/a	Forbes *et al.* 2016
Golden perch *Macquaria ambigua*	Billabong Creek, Vic	River	Rec	85% (79-100%)	4 x increase	Crook *et al.* 2016
Golden perch *Macquaria ambigua*	Edward River, Victoria	River	Rec	46% (27-65%)	Inconclusive	Crook *et al.* 2016

Species	Location	Environment	Fishery type	Contribution to fishery (%)	Change in fishery	Source
Golden perch *Macquaria ambigua*	Murrumbidgee River, Victoria	River	Rec	33% (29-37%)	4 x increase	Crook *et al.* 2016
Golden perch *Macquaria ambigua*	Reedy Lake, Australia	Reservoir	Rec	47 ± 9%	8 x increase	Hunt *et al.* 2010
Golden perch *Macquaria ambigua*	Kangaroo Lake, Australia	Reservoir	Rec	55 ± 9%	10 x increase	Hunt *et al.* 2010
Golden perch *Macquaria ambigua*	Lake Charm, Australia	Reservoir	Rec	90 ± 5%	4 x increase	Hunt *et al.* 2010
Golden perch *Macquaria ambigua*	Victoria, Australia	Rivers, Lakes, Reservoirs	Rec	22 ±7%	Likely increased overall	Ingram *et al.* 2015b
Golden perch *Macquaria ambigua*	Campaspe River, Australia	River	Rec	54 ±27%	Likely increased	Ingram *et al.* 2015b
Golden perch *Macquaria ambigua*	Kow Swamp, Australia	Lake	Rec	32 ± 26%	Inconclusive	Ingram *et al.* 2015b
Golden perch *Macquaria ambigua*	Loddon River, Australia	River	Rec	39 ± 17%	Likely increased	Ingram *et al.* 2015b
Golden perch *Macquaria ambigua*	Gouldburn River, Australia	River	Rec	11 ±11%	Unlikely to have signif. impact	Ingram *et al.* 2015b
Japanese flounder *Paralichthys olivaceus*	Miyako Bay, Japan	Coastal Bay	Com	33.6%	Added 22.7% value to total landings	Iwamoto *et al.* 1998

Species	Location	Environment	Fishery type	Contribution to fishery (%)	Change in fishery	Source
Japanese flounder *Paralichthys olivaceus*	Japan	Coastal bay	Com	10.6 ± 4.4%	n/a	Kitada 2020
Japanese flounder *Paralichthys olivaceus*	Iwate, Japan	Coastal bay	Com	28.8%	1.2 x increase	Okouchi *et al.* 1999
Japanese flounder *Paralichthys olivaceus*	All Japan	Coastal	Com	11.7 ± 11.0%	Significant increase	Kitada & Oshino 2006
Japanese flounder *Paralichthys olivaceus*	Japan	Coastal	Com	25-40%	n/a	Kitada 1999
Japanese flounder *Paralichthys olivaceus*	Hokkaido, Japan	Coastal	Com	11-23%	n/a	Ishino 1999
Japanese flounder *Paralichthys olivaceus*	Fukshima, Japan	Coastal	Com	20.1%	Additional 60 tonnes p.a.	Tomiyama *et al.* 2008
Japanese scallop *Mizuhopecten yessoensis*	Japan	Coastal bay	Com	76.6 ± 20.4%	n/a	Kitada 2020
Japanese Spanish mackerel *Scomberomorus niphonius*	Seto Inland Sea, Japan	Open coastal	Com	4-42%	Added 8.3-21.6 tonnes (¥19.52 million)	Yamazaki *et al.* 2007
Japanese Spanish mackerel *Scomberomorus niphonius*	Seto Inland Sea, Japan	Coastal	Com	2.2 ± 1.9%	n/a	Kitada 2020
Kuruma prawn *Penaeus japonicus*	Japan	Coastal	Com	13.4 ± 5.4%	n/a	Kitada 2020
Largemouth bass *Micropterus salmoides*	Florida, USA	Reservoir	Rec	8.2-15.2%	No impact	Thompson *et al.* 2016
Largemouth bass *Micropterus salmoides*	Oklahoma, USA	Reservoir	Rec,	74%	n/a	Boxrucker 1986

Species	Location	Environment	Fishery type	Contribution to fishery (%)	Change in fishery	Source
Largemouth bass *Micropterus salmoides*	Kentucky, USA	Reservoir	Rec	24.5%	4-5 x	Buynak & Mitchell 1999
Largemouth bass *Micropterus salmoides*	Texas, USA	Reservoir	Rec	5.4-14.9%	n/a	Buckmeier *et al.* 2003
Largemouth bass *Micropterus salmoides*	Tennessee, USA	Reservoir	Rec	<2%	No impact	Hoffman & Bettoli 2005
Masu salmon *Onchorynchus masou*	Hokkaido, Japan	Open coastal	Com	2.5–5.7%	n/a	Miyakoshi *et al.* 2004
Mulloway *Argyrosomus japonicus*	Khappinghat Creek, Australia	Intermittently closed estuary	Rec	0%	No impact	Taylor *et al.* 2009
Mulloway *Argyrosomus japonicus*	Swan Lake, Australia	Intermittently closed estuary	Rec	0%	No impact	Taylor *et al.* 2009
Mulloway *Argyrosomus japonicus*	Smith Lake, Australia	Intermittently closed estuary	Rec	Majority	21-30 x increase	Taylor *et al.* 2009
Mulloway *Argyrosomus japonicus*	Georges and Richmond Rivers, Australia	Open estuary	Rec	7%	n/a	Taylor *et al.* 2021
Murray cod *Maccullochella peelii*	Murray River, NSW	River	Rec	7%	n/a	Forbes *et al.* 2016
Murray cod *Maccullochella peelii*	Murrumbidgee River, NSW	River	Rec	15%	n/a	Forbes *et al.* 2016

Species	Location	Environment	Fishery type	Contribution to fishery (%)	Change in fishery	Source
Murray cod Macculochella peelii	Copeton Dam, NSW	Reservoir	Rec	94%	n/a	Forbes *et al.* 2016
Murray cod *Maccullochella peelii*	Victoria, Australia	Rivers, Lakes, Reserviors	Rec	50 ± 8%	n/a	Ingram *et al.* 2015b
Murray cod *Maccullochella peelii*	Campaspe River, Australia	River	Rec	100%	Unlikely to have signif. impact	Ingram *et al.* 2015b
Murray cod *Maccullochella peelii*	Gunbower Creek, Australia	River	Rec	62 ± 24%	Inconclusive	Ingram *et al.* 2015b
Murray cod *Maccullochella peelii*	Kow Swamp, Australia	Lake	Rec	47 ± 13%	Inconclusive	Ingram *et al.* 2015b
Murray cod *Maccullochella peelii*	Lake Eildon, Australia	Reservoir	Rec	98 ± 1%	Supports fishery	Ingram *et al.* 2015b
Murray cod *Maccullochella peelii*	Loddon River, Australia	River	Rec	22 ± 22%	Unlikely to have signif. impact	Ingram *et al.* 2015b
Murray cod *Maccullochella peelii*	Gouldburn River, Australia	River	Rec	11 ± 6%	Likely increased	Ingram *et al.* 2015b
Murray cod *Maccullochella peelii*	Edwood-Wakool system, Australia	River	Rec	6% (0-67%)	n/a	Thiem *et al.* 2016
Murray cod *Maccullochella peelii*	Loddon River, Australia	River	Rec	50%	increased	Hall & Douglas 2008

Species	Location	Environment	Fishery type	Contribution to fishery (%)	Change in fishery	Source
Pacific threadfin *Polydactylus sexfilis*	Hawaii, USA	Estuary	Rec	8.7% total 71% local	1.1 x increase	Friedlander & Ziemann 2003 Ziemann 2004
Pink salmon *Oncorhynchus gorbuscha*	Japan	Open coastal	Com	16.6-26.4%	n/a	Ohnuki *et al.* 2015
Rainbow trout *Oncorhynchus mykiss*	Lake Eucumbene, Australia	Reservoir	Rec	17%	n/a	Faragher *et al.* 2007 Forbes *et al.* 2017
Red drum *Sciaenops ocellatus*	Cedar Lakes, Texas	Estuary	Rec	20%	1.22 x increase	McEachron *et al.* 1998
Red drum *Sciaenops ocellatus*	Texas, USA	Estuary	Rec	20%	2 x abundance 1.18 x rec catch	McEachron & Fuls 1996 Warren *et al.* 1994
Red sea bream *Pagrus major*	Kanagawa, Japan	Coastal bay	Com	14%	n/a	Ungson *et al.* 1993
Red sea bream *Pagrus major*	Japan	Coastal bay	Com	6.5 ± 1.8%	n/a	Kitada 2020
Red sea bream *Pagrus major*	Kagoshima, Japan	Coastal bay	Com	64-83%	n/a	Kitada 1999
Red sea bream *Pagrus major*	All Japan	Coastal bay	Com	9.5 ± 8.3%	Significant increase	Kitada & Oshino 2006
Salmon spp. *Oncorhynchus spp.*	Alaska, USA	River	Com, Rec	32%	n/a	Knapp *et al.* 2007
Sand whiting *Sillago ciliata*	Maroochy River, Queensland	Estuary	Com Rec	52% 44%	Inconclusive due to fish kills	Butcher *et al.* 2000

Species	Location	Environment	Fishery type	Contribution to fishery (%)	Change in fishery	Source
Short-spined sea urchin *Strongylocentrotus intermedius*	Hokkaido, Japan	Open coastal	Com	62-80%	n/a	Sakai *et al.* 2004
Striped bass *Morone saxatilis*	Chesapeake Bay, USA	Estuary	Rec	27% (5-38%)	n/a	Secor & Houde 1998
Swimming crab *Portunus trituberculatus*	Japan	Coastal bay	Com	19.5 ± 5.0%	n/a	Kitada 2020
Turbot *Psetta maxima*	North Zealand, Denmark	Open coastal	Com	3.7%	n/a	Støttrup *et al.* 2002
Whitefish *Coregonus laveretus*	Lake Constance, Switzerland	Reservoir	Rec	62 ± 5%	n/a	Eckman *et al.* 2007
Yellow perch *Perca flavascens*	South Dakota, USA	Reservoir	Rec	26% (5-41%)	n/a	Brown & Sauver 2002

Appendix B – Cost benefit analysis of stocking projects

Table 28 Cost-benefit analysis results from past fish stocking projects in the original study denomination.

Species	Country	Source	Analysis method	Currency	BCR	Equivalent annual value	NPV	Comments
Arctic char *Salvinus alpinus*	Norway	Moksness *et al.* 1998	NPV	USD 1998	n/a	n/a	-0.86 to -1.09 million	Small smolt (60-70g) Over 5 years Discount rate 10%
Arctic char *Salvinus alpinus*	Norway	Moksness *et al.* 1998	NPV	USD 1998	n/a	n/a	-1.00 to -1.46 million	Large smolt (300g) Over 3 years Discount rate 10%
Atlantic cod *Gadus morhua*	Norway	Moskness & Stole 1997	NPV	USD 1997	n/a	n/a	-262,061	Over 3 years Discount rate 15%
Atlantic salmon *Salmo salar*	Norway	Moksness *et al.* 1998	NPV	USD 1998	n/a	n/a	-0.57 to -0.70 million	Over 4 years Discount rate 10%
Barramundi *Lates calcarifer*	Tinaroo Dam, Australia	Rutledge *et al.* 1990	BCA	AUD 1990	32 52	285,600 2,499,375	n/a	Single year 1st values – no multiplier 2nd values used 3.18 multiplier of travel cost to estimate indirect and induced benefit values
Barramundi *Lates calcarifer*	Australia	Hamlyn & Beattie 1993	NPV	n/a	18	n/a	n/a	Single year
Black sea bream *Acanthopagrus schlegeli*	Taiwan	Liao & Liao 2002	BCA	NT 1999	1.001	n/a	19,328	Single year <1% recap needed to cover stocking cost

Species	Country	Source	Analysis method	Currency	BCR	Equivalent annual value	NPV	Comments
Brown marbled grouper *Epinephalus fuscoguttatus*	Indonesia	Yulianto *et al.* 2019	BCA	IDR 2018	3.55-4.82 1.19-1.99	28.4-42.0 million 17.8-29.9 million	n/a	Single stocking event of 1.2 yr Facilities costs not included
Brown trout *Salmo trutta* Rainbow trout *Oncorhynchus mykiss* Chinook salmon *Oncorhynchus tshawytscha*	Purrumbete Lake, Australia	Hunt *et al.* 2017	BCA	AUD 2014	4.8-16	0.41-1.42 million	n/a	Values varied depending upon the opportunity cost of travel time used (0-100% wage)
Chinook salmon *Oncorhynchus tshawytscha*	USA	Wahle *et al.* 1974	NPV	USD 1973	4.2	641,000-4,452,000	n/a	Harvesting costs not included
Coho salmon *Oncorhynchus kisutch*	USA	Wahle *et al.* 1974	NPV	USD 1973	6.6-7.4	7.28-7.78 million	n/a	Harvesting costs not included
Dusky flathead *Platycephaus fuscus* Sand whiting *Sillago ciliata*	Maroochy River, Australia	Bucher *et al.* 2000	Outlay model and Total Cost model	AUD 1998	0.07-0.09	n/a	-584,000 to -814,000	Over 5 years Compared models Influenced by fish kills
Freshwater salmonids *Oncorhynchus spp.*	Blue Mesa Lake, USA	Johnson & Walsh 1987	NPV	USD 1986	6.34	2.35 million	n/a	Single year Only annual comparison of rec value
Largemouth bass *Micropterus salmoides*	Taylorsville Lake, USA	Buynak & Mitchell 1999	BCA	USD 1995	3.9	0.412 million	2.06 million	Averaged over 5 years

Species	Country	Source	Analysis method	Currency	BCR	Equivalent annual value	NPV	Comments
Japanese flounder *Paralichthys olivaceus*	Hokkaido, Japan	Sproul & Tominaga 1992	BCA	Yen 1990	3.15	n/a	270 million	20 yr program timeframe Discount rate 8% Facility construction costs not included
Mongolian redfin *Culter mongolicus*	China	Lin *et al.* 2021	BCA	n/a	2.6	n/a	n/a	Single year
Pink salmon *Oncorhynchus gorbuscha*	Canada	Boyce *et al.* 1993	BCA	1992 USD	n/a	-16.1 million	n/a	Over 30 years Undiscounted State benefits and costs only No recreational or subsistence values included
Rainbow trout *Oncorhynchus mykiss*	Missouri, USA	Weithman & Haas 1982	TCM Income multiplier	USD 1980	7.1 22	2.9 million 9.9 million	n/a	Compared valuation techniques
Red drum *Sciaenops ocellatus*	Texas, USA	Rutledge *et al. 1989*	BEA	USD 1989	52-260	176.8 million	n/a	BCR range depends on projected post-release survival (1% or 5%)
Red drum *Sciaenops ocellatus*	South Carolina, USA	Rhodes *et al.* 2018	BCA	USD 2005	4.65	1.764 million	14.6 million	Over 10 years Discount rate 3.5%
Red sea bream *Pagrus major*	Kagoshima Bay Japan	Ungson *et al.* 2003	BCA	Yen 1990	1.65	327 million	n/a	Supplied fingerlings for stocking and aquaculture
Salmon spp.	British Columbia, Canada	Pearse 1994	BCA	USD 1993	0.61	n/a	-592 million	Life of program Discount rate 8% Facilities cost were irretrievable leading to negative result

Species	Country	Source	Analysis method	Currency	BCR	Equivalent annual value	NPV	Comments
Salmon spp.	British Columbia, Canada	Pearse 1994	BCA	USD 1993	1.6	n/a	165.3 million	Over 24 years 1993-2017) Discount rate 8% Facilities costs are foregone, so net benefits in continuing program
Sockeye salmon *Oncorhynchus nerka*	Canada	Boyce *et al.* 1993	BCA	USD 1992	n/a	-$12.0 million	n/a	Over 30 years Undiscounted State benefits and costs only No recreational or subsistence values included
Striped bass *Morone saxatilis*	North Carolina, USA	Patrick *et al.* 2006	BCA	n/a	0.082-0.170	n/a	n/a	Over 15 years Not including non-use values
Brown tiger prawns *Penaeus esculentus*	Exmouth Gulf, Australia	Longeragan *et al.* 2004	BCA	AUD 2003	0.96	-170,000	n/a	Single year Estimated positive return only 48% of time No capital costs
Brown tiger prawns *Penaeus esculentus*	Exmouth Gulf, Australia	Ye *et al.* 2005	BCA	AUD 2004	1.33	220,000	n/a	Single year Estimated positive return only 65% of time No capital costs
Chinese white shrimp *Penaeus chinensis*	Zhejiang China	Xu *et al.* 1997	NPV	n/a	5.2	n/a	n/a	Unclear what costs were included

Species	Country	Source	Analysis method	Currency	BCR	Equivalent annual value	NPV	Comments
Chinese white shrimp *Penaeus chinensis*	Xiamen Bay, China	Wang *et al.* 2006	NPV	n/a	7-10	n/a	n/a	Unclear what costs were included
Eastern king prawn *Melicertus plebejus*	Wallagoot Lake, Australia	Taylor 2017	NPV	AUD 2016	5.48	278,050	n/a	Used 2.3 multiplier for indirect and induced benefits
European lobster *Homarus gammarus*	Norway	Moksness *et al.* 1998	NPV	USD 1998	n/a	n/a	-7,000 to -10,000	Over 13 years Discount rate 10%
Greenlip abalone* *Haliotis laevigata*	Augusta, Australia	Hart and Strain 2016	NPV	AUD 2013	1.7	2.1 million	32 million	Discount rate 6%
Saucer scallop *Amusium balloti*	Australia	Dredge *et al 2002*	BCA	AUD 2000	2.65	1.03 million	10.2 million	Over 20 years Discount rate 8%
Short-spined sea urchin *Strongylocentrotus intermedius*	Hokkaido, Japan	Sakai *et al.* 2004	NPV	Yen 2000	2.05-3.45	3.96-5.85 million	n/a	Facilities costs not included

* Model predictions only

Appendix C – Economic value of SIPS impoundments

Table 29 Economic value of stocked impoundments in the Queensland SIPS program. Adapted from Gregg and Rolfe (2013). The revised values were based on comparison between the findings of Greg and Rolfe (2013) with the more detailed analysis by Rolfe and Prayaga (2007) at the same dams and then conversion of these values to AUD 2021.

Impoundment	Daily use value ($/day)	Conservative annual value ($ million/year)	Revised annual value ($ million/year)	Revised annual value (AUD 2021 million/year)
Bjelke-Petersen	$184.23	$1.35	$2.28	$2.62
Boondooma	$184.23	$2.26	$3.82	$4.39
Borumba	$184.23	$3.36	$5.67	$6.53
Burdekin Falls	$184.23	$0.20	$0.34	$0.39
Callide	$54.05	$0.19	$0.32	$0.37
Cania	$158.73	$0.61	$1.03	$1.19
Connolly	$66.23	$0.36	$0.61	$0.70
Cooby	$65.36	$0.12	$0.20	$0.23
Coolmundra	$97.09	$0.29	$0.49	$0.56
Cressbrook	$333.33	$4.18	$7.06	$8.13
Eungulla	$416.67	$1.59	$2.68	$3.09
Fairbairn	$344.83	$1.69	$2.85	$3.29
Glenlyon	$312.50	$5.70	$9.62	$11.08
Gordonbrook	$184.23	$0.10	$0.17	$0.19
Kinchant	$238.10	$1.10	$1.86	$2.14
Lake Dyer	$50.25	$0.53	$0.89	$1.03
Lake Gregory	$62.89	$0.12	$0.20	$0.23
Lake McDonald	$142.86	$0.31	$0.52	$0.60
Lake Monduran	$175.44	$2.29	$3.87	$4.45
Leslie	$344.83	$3.38	$5.71	$6.57
Maroon	$158.73	$1.25	$2.11	$2.43
Moogerah	$100.00	$0.87	$1.47	$1.69
North Pine Dam	$112.36	$2.91	$4.91	$5.66
Peter Faust	$196.08	$0.59	$1.00	$1.15
Somerset	$344.83	$10.42	$17.59	$20.26
Storm King Dam	$80.65	$0.18	$0.30	$0.35
Teemburra	$184.23	$0.15	$0.25	$0.29
Theresa Creek	$184.23	$0.60	$1.01	$1.17
Tinaroo Falls	$256.41	$7.05	$11.90	$13.71
Wivenhoe	$105.26	$2.18	$3.68	$4.24
Wuruma	$204.08	$0.50	$0.84	$0.97
Total		**$56.43**	**$95.28**	**$109.71**