Marine Biodiversity Hub

National Environmental Science Programme

# The role of restoration in conserving matters of national environmental significance in marine and coastal environments

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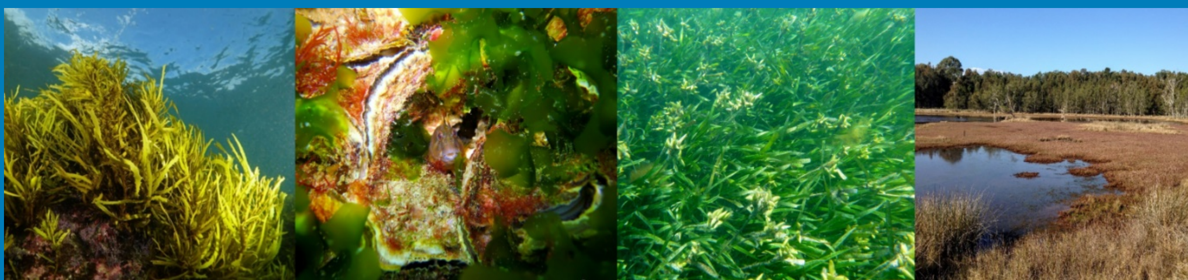
*Project E5 – The role of restoration in conserving Matters of National Environmental Significance*

16 December 2018 *Milestone 3 - Research Plan v4 (2018)*



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## Preferred Citation

McLeod I. M., Boström-Einarsson L., Johnson C., Kendrick G., Layton C., Rogers A. A., Statton J. (2019). The role of restoration for conserving Matters of National Environmental Significance. Report to the National Environmental Science Programme, Marine Biodiversity Hub. 186 pages.

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## Acknowledgement

This work was undertaken for the Marine Biodiversity Hub, a collaborative partnership supported through funding from the Australian Government's National Environmental Science Programme (NESP). NESP Marine Biodiversity Hub partners include the University of Tasmania; CSIRO, Geoscience Australia, Australian Institute of Marine Science, Museums Victoria, Charles Darwin University, the University of Western Australia, Integrated Marine Observing System, NSW Office of Environment and Heritage, NSW Department of Primary Industries. We would like to thank Dr James Fitzsimons and Dr Chris Gillies from The Nature Conservancy for helping to shape this project, Blanche D'Anatas for providing editorial support, Francisco Baena for providing fish data for the shellfish chapter, all the case study authors, Vishnu Prahalad from the University of Tasmania for editorial advice for the saltmarsh chapter, and the 60 attendees at the Marine and Coastal Habitat Workshop in Canberra which provided context and background information for this report.

# Important Disclaimer

# Contents

# List of Figures

# List of Tables

# EXECUTIVE SUMMARY

Healthy coastal habitats like seagrass meadows, coastal saltmarsh, kelp forests, coral and shellfish reefs, and mangrove forests ('blue infrastructure') are essential to the economic and social well-being of coastal communities. These habitats drive coastal productivity supporting our fisheries and other industries associated with recreation in marine environments, improve water quality, sequester carbon, protect shorelines from erosion, and support thriving biodiversity, including threatened species. These habitats are under pressure from coastal development, climate change, pollution, invasive species and other anthropogenic pressures, which have led to drastic declines in many of our important marine and coastal habitats.

Under the division of powers between the Australian Government and the states under the Australian Constitution, states and territories have the primary responsibility for environmental protection of coastal habitats within three nautical miles of the coastline. The *Environment Protection and Biodiversity Conservation Act* 1999 (C'th) (the EPBC Act) enables the Australian Federal Government to join with the states and territories in providing a national scheme of environment and heritage protection and biodiversity conservation. The EPBC Act focuses Australian Government interests on the protection of nine Matters of National Environmental Significance (MNES). These include World Heritage Areas and Ramsar wetlands, threatened and endangered species and habitats, and migratory species protected through international agreements, and Commonwealth Marine Areas.

Given the current state of decline in natural ecosystems, there is a general consensus that there are two paths to conserve critical habitats; habitats can either be protected from extractive or destructive human influences (e.g. through national parks, marine reserves, fishery closures, gear restrictions or riparian conservation), and/or actively rehabilitated towards a preferred healthy state (i.e. restoration). Early environmental conservation was primarily focused on the former of these methods, with the establishment of national parks and conservation areas globally, and sector-based management of remaining pressures. However, despite these intensive interventions, many habitats have continued to decline over the past half century. There is increasing recognition that protection by itself is no longer sufficient and interest and demand for rehabilitation in the form of interventions and restoration has been growing. Restoration is now seen as a key element in achieving conservation and environmental management goals internationally. In recent decades, nations such as the United States, Canada and the United Kingdom have embraced the need for large-scale marine and coastal restoration. Further, restoration also produces economic benefits. For example, restoration activities were recently estimated to contribute almost US$25 billion and 221,000 jobs annually to the United States economy.

In this report we review the state of four ecologically critical coastal marine habitats in Australia; seagrass meadows, kelp forests, shellfish reefs, and coastal saltmarsh wetlands, and evaluate (1) the Commonwealth responsibility for the habitat under the EPBC Act, (2) capacity of habitat restoration to insulate against loss and degradation of MNES, through restoration of key habitats and the species they support, (3) recent advances in restoration with the potential to improve outcomes associated with MNES.

This report demonstrates that each of the four habitats fall under up to six of the nine MNES, by being directly listed as or supporting threatened species or ecosystems, providing habitat for listed migratory species, and being important components of World Heritage Areas, Commonwealth waters, the Great Barrier Reef Marine Park, and Ramsar wetlands. For example, giant kelp (*Macrocystis pyrifera*) forests are listed as an endangered ecological community; temperate and subtropical saltmarshes are listed as a vulnerable ecological community and three saltmarsh species are listed as vulnerable. In addition, the habitats formed by the two primary reef-forming oyster species are under consideration for listing as endangered ecological communities under the EPBC Act. Coastal saltmarshes provide critical habitat for listed threatened species, such as the green and golden bell frog (*Litoria aurea*) and the orange-bellied parrot (*Neophema chrysogaster*), and migratory species such as the eastern curlew (*Numenius madagascariensi*s), the Pacific golden plover (*Pluvialis fulva*), the sharp-tailed sandpiper (*Calidris acuminata*), and the red-necked stint (*Calidris ruficollis).* Seagrass habitats make up a large proportion of the Great Barrier Reef Marine Park and World Heritage Area and support listed turtle species and dugong. Similarly, kelp forests support a disproportionately high number of endemic species, including several listed under the EPBC At, including the spotted handfish (*Brachionichthys hirsutus*, critically endangered), red handfish (*Thymichthys politus,* critically endangered), Ziebell's handfish (*Brachiopsilus ziebelli*, vulnerable), black rockcod (*Epinephelus daemelii*, vulnerable) and members of the Syngnathidae family (seadragons, seahorses and pipefish).

In Australia, marine and coastal habitat restoration began in the 1970s with seagrass transplantation trials in West Australia. Coastal wetland restoration, including for saltmarsh habitats, began in New South Wales in the 1990s with projects focused on restoring natural tidal exchange, excluding cattle, and controlling weeds. Recently there has been increasing recognition of both the value and the decline of habitats, prompting scores of restoration projects with expanding scope and scale. Kelp restoration trials began in the early 2000s with transplantation of endangered giant kelp in Tasmania. Shellfish reef restoration is a new activity in Australia with the first trial projects starting in 2014, along with an increase in related research into the function and structure of shellfish reefs. Recent research has focused on incorporating future climate scenarios and disease resistance into restoration planning.

Given that ecological restoration is a relatively new endeavour in many ecosystems across Australia, success has been varied. For example, despite decades of restoration practice on Australian seagrass species, our ability to improve the survivorship of outplants still remains relatively variable. Of the four habitats described in this report, saltmarsh restoration as a component of coastal wetland restoration appears to be the most advanced. Coastal wetland restoration success has been described at the scale of 100s of hectares, with plans to expand projects to 1000s of hectares in the near future. While early giant kelp restoration projects had inconclusive results, Operation Crayweed, which began in 2012 and aims to restore crayweed (*Phyllospora comosa*) forests to metropolitan Sydney has had some promising results. Outplanted crayweed patches rapidly became self-sustaining and expanded. Shellfish reef restoration has scaled up rapidly with two projects focused on building native flat oyster reefs in South Australia and Victoria at the scale of 10s of hectares, with plans to build 100s of hectares of reefs in the near future. It is too early to assess the long-term success of these reefs, but initial results are promising with high survival of shellfish and local recruitment.

While Australian restoration may be a relatively new initiative, we can benefit from a wealth of information from terrestrial and freshwater projects in Australia and from terrestrial, freshwater and coastal projects overseas. For example, thousands of hectares of seagrass have been successfully restored in Virginia in the US, and hundreds of hectares of shellfish reefs have benefited decades of restoration in Chesapeake Bay, US. With decades of experience from overseas to draw on, and with recent advances in restoration ecology, Australia has an opportunity to avoid the common growing pains experienced elsewhere. In general, restoration projects in most ecosystems overseas have suffered from a range of common problems and issues, including: (1) a lack of appropriate monitoring and reporting of the outcomes of projects, which prohibits learning and encourages repetition of past mistakes; (2) a general lack of pre-established and standardised guidelines and decision-support tools leading to poorly designed projects, with a higher risk of failure; and (3) a hesitance to support large-scale and long-term projects, instead favouring pilot-projects which may be less likely to succeed due to a mismatch between the scale (both temporal and spatial) of the stressor and ecological processes within the system, and the scale of restored habitat.

Restoration has not been a commonly used strategy within the context of MNES in coastal and marine areas. To date, most effort has focused on habitat protection and removal of stressors. Restoration has the potential to be a useful tool for managing Australia's valuable marine and coastal habitats, and provide coastal jobs and economic development opportunities. There are likely to be large benefits from national and regional leadership and knowledge sharing. The Department of Environment and Energy co-design process could be used as a central point of contact to connect potential partners and build capacity for more cross-sector partnerships and the National Environmental Science Program could continue to be strategically used to fund research associated with restoration knowledge gaps.

This report describes opportunities to include restoration in the toolkit of management actions for MNES and makes the following recommendations:

**Recommendations**

### *Consider all options for the recovery of MNES*

- *Consider all potential management actions for the recovery of MNES*. Consider threat removal, habitat protection (reserves) and active interventions, and weigh up potential costs and benefits when considering management actions to preserve MNES.
- *Consider restoration as a complementary tool to other management actions*. Restoration does not necessarily need to be undertaken sequentially or separately from other

management activates, or only as an offset requirement. Multiple management actions can complement each other and may result in positive feedback loops.

### *Consider the risk of inaction, as well as action*

- *The risk of action should be weighed against the risk of inaction.* Currently, conservation is often viewed through a lens of preservation or protection of pristine systems, where any habitat-modifying intervention to that system is considered a risk. However, given that all four ecosystems in this report have experienced substantial declines under current management strategies, it is clear that novel solutions should be considered. Risk should clearly be taken into account when considering restoration actions, however the risk of not taking restorative actions should also be considered in these assessments.

### *Develop a policy pathway to restoration*

- *Modify the interpretation of existing policy or develop fit-for-purpose policy to distinguish restoration from development*. As a relatively new initiative in Australia, ecological restoration does not have a history of targeted policy in the marine and coastal environment. Any proposed restoration activity is therefore judged based on policy that may be a poor fit for the activity. For example, natural habitat provision for shellfish or kelp restoration is regulated by the *Environmental Protection (Sea Dumping) Act* 1981 (C'th)). In some circumstances only a shift in definition or permit process is needed for this change, however, new policy may need to be developed in the medium to long-term.
- *Streamline permitting for marine and coastal habitat restoration projects.* In contrast to terrestrial restoration projects, marine and coastal restoration projects almost always occur on Crown land, therefore, governments (at a Federal, state or local level) are required to be more involved in and have more of an interest in the proposed restoration activities. In addition, permit proponents need to consider other regulatory, insurance and safety issues such as working in or near water (e.g. diving and marine biosecurity protocols). These factors along with permitting processes that are not fit-for-purpose mean that timelines from conception to implementation are therefore generally much slower compared to terrestrial projects, which places a heavier financial burden on proponent.
- *Enable permitting processes to weight overall benefits, costs and risk*. Permitting processes and culture could be refined to weigh the overall potential benefits of a project with the risk of small-scale damage. For example, it is very difficult to get a permit for shellfish reef restoration if there is seagrass present, even if the seagrass is in poor condition and located in an area with evidence of historical shellfish reefs.
- *Use permit process to ensure best practice procedures*. Encourage appropriate planning and monitoring to ensure best practice ecological restoration by including appropriate requirements in the permit approvals process. These could include ensuring restoration actions are well matched with stated objectives and requiring appropriate monitoring and reporting on the progress and outcomes of projects.
- *Enable new funding opportunities for restoration*. Develop and maintain pathways to support restoration projects through offsets, environmental insurance, private-public

partnerships, and community led volunteer projects, and co-investment states and local government.

### *Value ecosystem services of blue infrastructure.*

- *Prioritise research to estimate the economic value of habitats.* Marine and coastal habitats can provide numerous benefits such as supporting fish productivity, carbon sequestration, nutrient cycling, coastal protection and recreation. Decision makers need to be able to weigh up the relative costs and benefits of coastal development, protection or restoration and to do this they need robust, accessible and defensible data on the ecological function and economic value of habitats and the ecosystem services they provide. If no quantified value is available, the risk is that the value of ecosystem services are unlikely to be included in decisions.

### *Consider recent history and plan for a changing climate*

- *Historical assessments should be included when setting baselines for protection-focused management.* If historical baselines are not defined permitting processes may then defend the status quo rather than other historical or desired states.
- *Challenge the assumption that protected areas are pristine.* It is a common presumption that protected areas are in a pristine condition, and therefore not appropriate sites for restoration. However, protected areas where some stressors are reduced may be ideal areas for restoration, and restoration may be needed to preserve the value of protected areas.
- *Challenge the assumption that restoration will restore areas to being pristine*. In most cases it will not be possible to return to a pre-impact 'pristine' state. Many restoration projects now focus on restoring critical ecosystem function and services. For example a restored kelp forest may not be identical to the historical state, but is likely to support a more productive and biodiversity system than the urchin barren it replaced.
- *Include climate change predictions into restoration planning*. For example, saltmarshes are vulnerable to sea level change, and restored saltmarshes at current locations are likely to be inundated in the near future. Space for new saltmarshes, at locations suitable in future conditions should therefore be included in restoration or management plans.

### *Invest in knowledge sharing, collaboration and best practice guidance*

- *Learn from international experience*. Work with other countries that have longer histories with restoration to learn from their experiences. Lessons from overseas could be used to inform policy, decision-making tools, workflows, and best practice guidelines in Australia.
- *Build on the Blue Carbon Initiative to include other habitats*. The Blue Carbon Initiative is a good example of an international initiative where Australia has provided leadership. This could be expanded to include other marine and coastal habitats such as kelp forests so that they can be included in plans to protect and restore marine and coastal ecosystems for their 'Blue Carbon' value.
- *Build capacity in partner nations for restoration of marine and coastal habitats.* Especially when these are linked to food security, alternative livelihoods, and shoreline protection.

This could be an important component of Australian foreign aid in the future that may be more cost-effective than investing in built infrastructure. The Australian Centre for International Agricultural Research has supported mangrove and coral restoration projects and this could provide a base for an expanded focus.

- *Support marine and coastal habitat restoration network.* Networks such as the Shellfish Reef Restoration Network, the Seagrass Restoration Network and the Australian Coastal Restoration Network may provide useful contacts to assist with development of national policy, recovery plans and disseminating best practices.

### *Consider building on the success of early projects by investing in larger projects to demonstrate success at scale*

- *Consider positive feedback loops, and system-wide restoration approaches*. Projects could target a variety of habitats, and the stressors causing their decline within a system rather than just addressing each habitat and threat separately. For example, water quality improvement through wastewater treatment upgrades could be matched with active habitat restoration. Also, there can be positive feedback loops between and within habitats. For example, oyster reef restoration can encourage the growth of seagrass meadows nearby, and healthy seagrass meadows are associated with less disease in nearby coral reefs.
- *Investing in larger projects to attract more co-investment.* Government investment is likely to encourage buy-in from a wider range of stakeholders and may attract funding from new sources. For example, the $990,000 investment into shellfish reef restoration in South Australia as 'natural infrastructure' help encourage co-investment for the >$4 million project.
- *Avoid spreading funding and effort too thin*. Small projects are important because they provide the research and development necessary for scale-up, and often include many community stakeholders. However, underfunding many small projects may not lead to success, and some types of restoration may only succeed at a larger scale.

# 1 INTRODUCTION

Healthy coastal habitats or 'blue infrastructure' such as seagrass meadows, coastal saltmarsh, kelp forests, coral and shellfish reefs, and mangrove forests are essential to the economic and social well-being of coastal communities (e.g. Kazmierczak and Carter 2010). These habitats drive coastal productivity supporting our fisheries and other industries associated with recreation in marine environments, improve water quality, sequester carbon, protect shorelines from erosion, and support thriving biodiversity, including threatened species (Barbier *et al.* 2011).

Coastal habitats and the benefits they provide are under pressure from development, climate change, pollution, invasive species and other anthropogenic pressures. This has led to drastic declines in many important marine and coastal habitats. For example, the cover of dense giant kelp forests is now <5% of that recorded in the 1970s, and recent heat waves have caused loss of kelp over 1000s of kilometres of coastline in Western Australia (Connell *et al.* 2008; Johnson *et al.* 2011; Wernberg *et al.* 2016); over 1,000km$^2$ of seagrass meadows have been lost from the Shark Bay World Heritage Area (Fraser *et al.* 2014; Thomson *et al.* 2015); native shellfish reefs are considered Australia's most imperilled coastal habitats with native flat oyster reefs at less than 1% of their former abundance (Gillies *et al.* 2018); the Great Barrier Reef has degraded through major bleaching events and cyclones (Hughes *et al.*, 2017); vast mangrove forests have been dying in the Northern Territory (Duke *et al.* 2017); and coastal saltmarsh have been listed as endangered in parts of New South Wales.

Under the division of powers between the Australian Government and the states under the Australian Constitution, the states and territories have the primary responsibility for environmental protection of coastal habitats within three nautical miles of the coastline. These state and territory waters contain most of the coastal habitats and thus the state and territory governments are generally responsible for their management. The Environment Protection and Biodiversity Conservation Act 1999 (the EPBC Act) enables the Australian Federal Government to join with the states and territories in providing a national scheme of environment and heritage protection and biodiversity conservation. The EPBC Act focuses Australian Government interests on the protection of nine Matters of National Environmental Significance (MNES), with the states and territories having responsibility for matters of state and local significance.

In response to these threats the Australian Government has listed giant kelp as an endangered ecological community and subtropical and temperate coastal saltmarsh wetlands as a vulnerable ecological community under the EPBC Act 1999 and many Ramsar Wetlands and World Heritage properties contain saltmarsh wetlands that are degraded and exposed to additional risk in a changing climate (Laegdsgaard 2006).

## 1.1 The Environmental Protection and Biodiversity Conservation Act 1999 (C'th)

The EPBC Act is the Australian Government's central piece of environmental legislation, administered by Department of the Environment and Energy (DoEE). It provides a legal framework to protect and manage nationally and internationally important flora, fauna, ecological communities and heritage places. These are defined in the EPBC Act as MNES.

The objectives of the EPBC Act are to:

- provide for the protection of the environment, especially MNES
- conserve Australian biodiversity
- provide a streamlined national environmental assessment and approvals process
- enhance the protection and management of important natural and cultural places
- control the international movement of plants and animals (wildlife), wildlife specimens and products made or derived from wildlife
- promote ecologically sustainable development through the conservation and ecologically sustainable use of natural resources
- recognise the role of Indigenous people in the conservation and ecologically sustainable use of Australia's biodiversity
- promote the use of Indigenous peoples' knowledge of biodiversity with the involvement of, and in cooperation with, the owners of the knowledge.

## 1.2 Matters of National Environmental Significance

The nine matters of national environmental significance are:

- world heritage properties
- national heritage places
- wetlands of international importance (often called 'Ramsar' wetlands after the international Ramsar Convention, under which such wetlands are listed)
- nationally threatened species and ecological communities
- migratory species
- Commonwealth marine areas
- the Great Barrier Reef Marine Park
- nuclear actions (including uranium mining)
- a water resource, in relation to coal seam gas development and large coal mining development.

In addition, the EPBC Act confers jurisdiction over actions that have a significant impact on the environment where the actions affect, or are taken on, Commonwealth land, or are carried out by a Commonwealth agency (even if that significant impact is not on one of the nine matters of 'national environmental significance').

Generally, the EPBC Act is enacted in two ways. Firstly, through the nomination of a native species, ecological community or threatening process for listing under the Act. The nomination is assessed by the Threatened Species Committee, and if a species, community or process is determined to be threatened, the Committee provides their assessment in the form of a 'conservation advice', which outlines the eligibility for listing and immediate conservation priorities. Conservation advice provides guidance on immediate recovery and threat abatement activities that can be undertaken to ensure the conservation of a newly listed species or ecological community. In some cases, the Minister for the Environment may make or adopt and implement recovery plans for threatened fauna, threatened flora (other than conservation dependent species) and threatened ecological communities listed under the EPBC Act. Recovery plans set out the research and management actions necessary to stop the decline of, and support the recovery of, listed threatened species or threatened ecological communities; however, they have no responsibility to enact the identified necessary management actions. Secondly, the EPBC Act is enacted when an action or activity could have significant impact on a MNES. In this situation, the project will be assessed by the DoEE, which then makes a recommendation to the minister for the environment about whether or not the project should be approved to proceed, or referred for further assessment.

Given the current state of decline in global ecosystems, there is a general consensus that there are two paths to conserve critical habitats; habitats can either be protected from extractive or destructive human influences (i.e. national parks or marine reserves), and/or actively rehabilitated towards a preferred healthy state (i.e. restoration). Early environmental conservation was primarily focused on the former of these methods, with the establishment of national parks and conservation areas globally (Jordan and Lubick 2011). However, in the past half century, interest and demand for the latter approach in the form of interventions and restoration has been growing. Restoration is now seen as a key element in achieving conservation and environmental management goals internationally. Restoration is considered an appropriate management action when natural ecosystem recovery is restricted by physical modification of the coast, lack of recruitment, local extinction, or when species dependent on the coastal habitats are facing local extinction due to habitat loss (Perrow and Davy 2002).

## 1.3 What is restoration?

The Society for Ecological Restoration International Science & Policy Working Group (2004) defines restoration as the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed. Ultimately, restoration attempts to return an ecosystem to its historic trajectory. However, in the context of a long-history of human modification and a changing climate reaching historical baselines may not be possible. The terms ‘repair’ or ‘rehabilitation’ have been suggested and used when reaching a historic baseline is not possible, and the target is instead to replace the structural or functional characteristics of an ecosystem that have been diminished or lost (Perrow and Davy 2002). In this report, we define restoration as including the definitions of repair and rehabilitation.

Active restoration is where management techniques such as transplantation, planting seeds or seedlings, or the construction of artificial habitats with their natural range are implemented, while passive restoration focuses on removing the impact of environmental stressors such as pollution or poor water quality, which prevent natural recovery of the ecosystems occurring

(Perrow and Davy 2002). The term intervention, referring to any human-driven environmental management action is now being used more frequently used both in ecology and in the broader field of environmental science in recent years (Hobbs *et al.* 2011).

## 1.4 History of ecological restoration

Ecological restoration, as we define it now, started in the early 1930's, as the first projects focusing on restoration of an ecosystem rather than just creating specific functions were conducted (Jordan and Lubick 2011). These early projects were driven by enthusiastic individuals, but failed to gain acceptance amongst ecologists more generally because it was generally assumed that habitats would recover on their own. However, acceptance and demand for restoration gained momentum following increased awareness of environmental degradation highlighted in books such a *Silent Spring* in 1962, which documented the adverse effects on the environment of the indiscriminate use of pesticides and birthed the environmentalist movement (Murphy 2007). In terrestrial systems, this momentum continued to grow in the 1960's, where the restoration of prairie and grassland ecosystems dominated the restoration scene in the American Midwest (Jordan and Lubick 2011). In the following decades ecological restoration branched out to a range of terrestrial ecosystems (Martin 2017), but has been largely focused on pine forests (Moore *et al.* 1999; Allen *et al.* 2002; Halme *et al.* 2013; Martin 2017 *et al.*), and grass and heathlands (Anderson 1995).

The restoration of wetlands was pioneered in the early 20th century, however similar to terrestrial systems, projects' objectives focused on recreating services provided by the ecosystems, rather than the ecosystems themselves. For example, there are records from as early as the late 1800's of replanting mangrove and saltmarsh species over large areas (sometimes up to 100s of hectares) in Australia, China, Europe, New Zealand, and the United States. The main objectives of these efforts were to slow erosion, reduce channel siltation, and reclaim land for agriculture (Craft *et al.* 2008). However, many of these early projects introduced new species, which have caused ongoing problems. For example, rice grass (*Spartina anglica*) was introduced to Australia based on its value to coastal engineering and agriculture but is now an invasive species that requires ongoing management (Kriwoken and Hedge, 2000). In the 1970's and 1980's however, awareness of the importance of mangroves and wetlands was growing, and the restoration of these systems for their own sake was increasingly recognised as a valid intervention (Jordan and Lubick 2011).

In recent decades, nations such as the United States, Canada and the United Kingdom have embraced the need for large-scale marine and coastal restoration (Gillies *et al.* 2015). The restoration economy (including terrestrial and freshwater projects) was recently estimated to contribute almost US$25 billion and 221,000 jobs annually to the US economy (BenDor *et al.* 2015). In 1996, the United States National Oceanic and Atmospheric Administration (NOAA) created the Community-based Restoration Program, overseen by the NOAA Restoration Centre. This program provided funding and technical support for thousands of projects around the United States. In 2000, the United States Congress created a federal interagency Estuary Habitat Council to support 'mid-sized' restoration projects (over US$1,000,000) and the *American Recovery and Reinvestment Action Act* 2009 (USA), provided $167 million for mid-scale restoration projects. Much of the funding for restoration in the US has come from disaster litigation. For example, as part of British Petroleum's Deep Water Horizon disaster mitigation the company will pay US$100 million to the North American Wetlands Conservation Fund for the purpose of wetlands restoration and conservation projects and US$2.4 billion to the National Fish and Wildlife Foundation, much of which to is targeted to restoration activities (The Environmental Law Institute & Tulane Institute on Water Resources Law & Policy 2014).

Australia's commitment to restoration is incorporated into international agreements. Australia has ratified commitments under the 2011 Bonn Challenge (Government of Germany & International Union for the Conservation of Nature 2011), which has the goal of restoring 150 million hectares and incorporates the goals of the 2010 Aichi targets under the Convention on Biological Diversity (International Union for the Conservation of Nature 1993). This includes restoring at least 15% of degraded ecosystems globally, through a commitment to restore 1000km$^2$ 'of fragmented landscapes and aquatic systems [...] to improve ecological connectivity' by 2015 (see Target 5, National Targets), under the New York Declaration on Forests, which highlights restoration of degraded ecosystems as a critical conservation tool in the Anthropocene, was signed in 2014 (United Nations 2014 and see Suding *et al.*, 2015). Further, Australia includes restoration as a key component of conservation of biodiversity in the natural environment (Natural Resource Management Ministerial Council 2010).

## 1.5 Ecological restoration in Australia

Australian ecosystems have been managed and shaped by Aboriginal peoples for tens of thousands of years. Ecological restoration has emerged in response to the reduction in quality and quantity of native flora and fauna that has occurred since Colonial settlement of Australia in the late 18th century (McDonald and Williams 2009). The history of ecological restoration in Australia has followed a similar trajectory to other locations, with new ecosystems (where restoration is undertaken) added in order of accessibility (to practitioners) and visibility (of problems and restoration outcomes). The first restoration trials in the 1920s and 1930s focused on terrestrial ecosystems including fire-adapted sclerophyll communities, desert vegetation, rainforests and grasslands or low shrublands (McDonald and Williams 2009). These trials were followed in the mid-1940s to 1960s by more widespread efforts to raise environmental practices, and soil stabilisation and revegetation work was carried out by the agricultural, water supply and resource extraction sectors (McDonald and Williams 2009). The urban bushland 'regeneration' movement started in Sydney in the late 1960s in response to growing public environmental consciousness. This saw the development of a minimal-intervention approach to assist natural recovery of natural vegetation. In the 1980s, the growing community awareness of the value of natural habitat led to the development of non-government organisation (NGO) and extension programs such as Greening Australia, Land for Wildlife, Trust for Nature Victoria, Trees for Life and Landcare which mostly focused on the agricultural sector. This was followed by freshwater restoration projects of rivers and wetlands (Lake 2005) in the 1990's to 2000's, including the large-scale Murray-Darling River restoration project established in 2004 (Murray-Darling Basin Commission, 2004).

In the marine realm, much of Australia's conservation focus has up been on protection of large areas of ocean and shallow coastal habitats through the implementation of marine protected areas. For example, the Marine Bioregional Planning Program, implemented under the EPBC Act, resulted in the establishment of the Commonwealth marine resources network, with more than 2.3 million square kilometres added to the Commonwealth’s marine protected areas. While marine protected areas should limit destructive activities that threaten key habitats, this has not always led to the recovery of threatened species and habitats. Further, while all four Marine bioregional plans (north, north-west, south-west, and temperate east) and the South-east marine bioregion profile, list vulnerable species such as dolphins, turtles and dugongs as conservation values within their plans, and for some species highlight the loss of habitat as a critical threat, there is no mention of ecological restoration as a tool to protect these vulnerable species. In contrast, the National Marine Science Plan 2015-2015 (National Marine Science Committee 2015) included recommendations to ‘develop, test and apply methods to mitigate the impacts of coastal hazards, including eco-engineering and restoration approaches’ for better management of urban coastal areas. The Australian Government has developed Australian Ramsar management principals, and include restoration as an action to consider when developing management plans. Overall, there appears to be a lack of consensus on the role of restoration in the conservation and preservation of Australia’s coastal ecosystems, and a need for clearer integration into a national legislative framework.

Recently, attention has begun to focus on shallow intertidal coastal areas, including estuaries. In 2001 a national audit demonstrated that 29% of Australian estuaries are considered to be ‘extensively modified’, particularly those in the east, south-east and south-west (Northern Land and Water Resources Australia. 2002). Creighton (2013) aimed to develop a national plan for action to restore Australia’s estuaries in the Fisheries Research and Development Fund report ‘Revitalising Australia’s Estuaries: the business case for repairing coastal ecosystems to improve fisheries productivity, water quality, catchment hydrology, coastal biodiversity, flood control, carbon sequestration and foreshore buffering’. This proposed a $350 million investment in Australian estuary repair and argued that this investment would be paid back within five years through ecosystem services (Creighton *et al.* 2013; 2015). In the past few years attention has turned towards subtidal habitats such a shellfish and coral reefs, with the establishment of shellfish reef restoration trials in most states, coral restoration trials on the Great Barrier Reef, and the Federal Government’s 2018 announcement of AU$100 million for reef restoration research.

With recent developments in the field, there is a growing recognition that national coordination of Australian restoration efforts would be useful to develop best practice guidelines, pool resources and involve multiple stakeholders. While some effort has been made to establish these groups, they are often poorly funded, local in focus and/or focused on projects above the high tide mark. For example, Coastcare is an extension of the Landcare movement and comprises 2000 groups of community volunteers (Gillies *et al.* 2015). They are active in the coastal restoration space but largely focus on dune erosion, loss of native coastal plants and animals, stormwater pollution, weeds and control of human access to sensitive and vulnerable areas. Marine habitat restoration has largely been excluded from terrestrial habitat funding programs most often delivered through the natural resource management (NRM) agencies.

In 2014, the Australian Government recognised OceanWatch as the national organisation responsible for the delivery of its marine NRM related programs. OceanWatch has the stated goal of enhancing fish habitats and improving water quality in estuaries and coastal environments but they are a small, poorly resourced agency. Over the last few years, there has been an increase in restoration projects and trials, many led by The Nature Conservancy through their Great Southern Seascapes Program (Fitzsimons *et al.* 2015). Several communities of practice have been established focussing on national coordination and knowledge sharing for particular habitat types such as the Shellfish Reef Restoration Network and the Seagrass Restoration Network. In 2017, the Australian Coastal Restoration Network was formed with support from the NESP Marine Biodiversity Hub, NESP Tropical Water Quality Hub and The Nature Conservancy, to link these networks together and link the restoration community to organisations such as the Australian Marine Science Association, The Society for Ecological Restoration Australasia and The Coastal Society (McLeod *et al.* 2018a). Over the last few years, there has been a rapidly growing community awareness of and engagement with restoration. This has the potential to raise awareness of conservation issues more generally and encourage a range of stakeholders to work together towards positive action. Supporting national knowledge sharing and coordination is likely to lead to the continued expansion of this in Australia, potentially leading to a ‘restoration economy’ similar to what has been achieved in the US.

## 1.6 Considerations for restoration as a tool for the conservation of MNES

Conservation organisations often use both protection and restoration as tools to conserve biodiversity, and management guidelines advocate for the use of both (Possingham *et al.* 2015). In contrast, the EPBC Act has been enacted to protect the areas, species or communities from development activities, and restoration has generally not been considered as a valid tool for marine and coastal habitat management, but tree planning and mine site remediation are commonly components of land-based development project approvals. However, recent research suggests that conservation strategies that include both restoration and protection are more likely to achieve biodiversity conservation targets or the provision of ecosystem services, although outcomes can be highly context dependent. In some cases, such as highly degraded environments, restoration may actually be a preferred approach in terms of return on investment compared to protection (Possingham *et al.* 2015; Saunders *et al.* 2017).

The EPBC Act requires that impacts to MNES must be avoided, mitigated or offset (Bos *et al.* 2014), in order of priority. From the EPBC Act definition of each term, restoration would fall under the offset category of the EPBC Act (ten Kate *et al.* 2004). Following the mitigation hierarchy therefore places restoration as a last resort, which may explain why interventions have been rarely enacted under the act. However, mitigation of the likely impacts of a development through restoring and thus increasing the resilience of an ecological community or habitat to the pressures of development might be at least as effective. As restoration efforts would target the specific community or species to be impacted, this may address a key societal preference for the local application of offsets. Rogers and Burton (2016) showed that Australians preferred direct offsets such as improving degraded habitats to indirect activities such as a research program and were strongly against offsets in locations other than where the impact occurred.

### 1.6.1 How has the EPBC Act been implemented?

Activities that may trigger the EPBC Act, undergo a process of referral (proposals of activities), public comment and decision. A referral is an application for the approval of an action that could have a significant impact on any MNES. After receiving a referral, the Minister (or delegate) decides whether the action has, will have, or is likely to have a significant impact on a MNES. When interrogating the database of referrals we identified 10 referrals that were deemed as 'clearly unacceptable' and the proposals withdrawn. A further 12 referrals were not approved under the EPBC Act following a public consultation period (Appendix 1). Of these 22 instances where the EPBC Act was formally triggered, only four affected a marine ecosystem. The only exclusively marine proposal was to conduct three-dimensional seismic surveys in the Muiron islands marine management area (Ref: EPBC 2012/6680), which was assessed as 'clearly unacceptable' under the EPBC Act. Three were major development works involving large scale terrestrial and marine developments.

1. *Shoalwater Bay, Queensland* (Ref: EPBC 2008/4366): The project involved establishing a major new coal mine, railway and port to export coal for electricity production. It involved a port development in Shoalwater bay and Corio Bay Ramsar wetlands, an undeveloped part of the Queensland coastline, and was not allowed to proceed.
2. *Great Keppel Island, Queensland* (Ref: EPBC 2009/5095): The proposed development included a 300 room hotel and day spa, 1700 low rise tourism resort villas, 300 tourism resort apartments, a 560-berth marina, ferry terminal and yacht club, country club, retail village, 18-hole championship golf course and a sporting oval.
3. *Ralphs Bay, Tasmania* (Ref: 2006/3193): Mixed housing and marina development.

Number 2 and 3 above were eventually allowed to proceed. While the rejection of proposals is not the only way that the EPBC Act is enacted, and the mere existence of an Act is likely to prevent damage to the environments it protects, it seems clear that shallow coastal environments have not received a lot of attention under this Act. Partly this is due to the division of legislation of coastal waters, where states have responsibility for waters within three nautical miles of the coastline, and the Federal Government assuming responsibility for waters beyond up to 200 nautical miles). However, it is also likely that an historical undervaluing of these productive ecosystems is a contributing factor to their under-representation in matters relating to the EPBC Act.

Another pathway to action under the EPBC Act is through recovery plans. Recovery plans are for greatly diminished species or communities. Recovery plans for threatened species or ecosystems listed under the EPBC Act "*..should state what must be done to protect and restore important populations of threatened species and habitat, as well as how to manage and reduce threatening processes"* (DoEE 2018). To the best of our knowledge, there are no recovery plans in place for marine and coastal habitats listed as MNES, but these are listed as required for giant kelp and saltmarsh. Restoration could take place before this level of degradation is reached, because of the ecosystem provided by habitats. There are recovery plans approved for species that use marine and coastal habitats such as nurse sharks (*Carcharias taurus*), sea lions *(Neophoca cinerea*), fur seals (*Arctocephalus tropicalis*) and elephant seals (*Mirounga leonina*). These recovery plans suggest regulation of fishing practices and other pressures that affect these species, but most do not target a specific habitat. Exceptions include the Recovery Plan for Marine Turtles in Australia, which supports the Raine Island Recovery Project where critical island nesting habitat is being restored through beach reprofiling, and recovery plans for handfish that include replacing spawning habitat (see case study, Section 7). This focus on charismatic species was further highlighted in an analysis of how migratory marine species are protected by the EPBC Act, where the authors noted that the primary tool to conserve biodiversity in Australia was through the implementation of protected areas (Miller *et al.* 2018), rather than direct interventions.

The Threatened Species Commissioner and the Threatened Species Strategy are separate but complimentary to the EPBC Act. The Threatened Species Strategy suggests habitat restoration as a way to assist the recovery of threatened or endangered plant and bird species; it generally does not target marine habitats or species (Australian Government 2015). In contrast, the Australian Government has committed to plant 20 million trees to improve native vegetation and habitat that supports native species while contributing to reducing greenhouse gases. A target of the Threatened Species Strategy is that 80% of the 20 Million Trees projects support threatened species by providing suitable habitat. More than $30 million is being directed towards tree planting projects that have direct threatened species outcomes through restoring the extent, connectivity and condition of native habitat. Example species targeted through tree planting are the helmeted honeyeater and the Leadbeater's possum.

Given the importance of marine and coastal environments (outlined in detail in the following chapters), and the acceptance and increasing appetite for restoration in terrestrial systems, it begs the question: why has restoration in marine and coastal systems not been considered? First, the marine and coastal environment has been considered so vast and naturally resilient that it is likely to recover naturally, without intervention, as long as threats and stressors affecting the system are removed. This has prompted a focus on establishing marine reserves and protected areas, similar to how national parks where the premier conservation tools of terrestrial ecosystems in the early 1800's. Second, most of the candidate areas for restoration are within three nautical miles of the coastline and are therefore primarily management by the states. Third, marine and coastal areas are generally not privately owned, and coastal management is carried out by multiple levels of government, which are generally poorly coordinated (Clark and Johnston, 2017), so no individuals or groups, or governments may feel that they have direct responsibility for restoration them. Forth, because marine and coastal systems are largely 'out of sight', and therefore 'out of mind' for a majority of Australians, there is less awareness of the issues facing marine systems, and the current state of important habitats. Finally, working in marine and coastal systems is fraught with significant logistical challenges, making these interventions expensive and difficult to implement. Many marine and coastal restoration projects have failed in the past and this is likely to have decreased manager's confidence in restoration.

Given these challenges, should restoration be considered in the marine environment? The current state of marine biodiversity decline in shallow coastal marine ecosystems highlights that existing conservation tools are not sufficient to protect these vulnerable ecosystems. Marine Protected Areas (MPAs) on their own have proven insufficient to prevent declines in biodiversity (with some notable exceptions) as natural rates of recovery may be too slow or it may be impossible to reverse past degradation (Borja *et al.* 2010; Lotze *et al.* 2011; Graham *et al.* 2014). MPAs have become contentious in Australia and thus expensive to implement. Restoration may provide quicker results that passively waiting for natural recovery and may be supported by a wider group of stakeholders.

While marine and coastal restoration techniques are generally at early stages of scaling up, restoration in freshwater environments is substantially more advanced. Revegetating banks to reduce erosion, re-configuring channels to restore stream complexity and introducing snags and fish ladders to support fish migration are all accepted restoration techniques that have been implemented in Australia (Nicol *et al.* 2004; Brooks and Lake 2007) and overseas (Katz *et al.* 2007). While these systems are not analogous to marine and coastal systems in all aspects, this long history of restoration in terrestrial and freshwater systems should be used to inform restoration in estuarine and marine systems (Elliott *et al.* 2007). Marine and coastal restoration has the opportunity to learn from and mitigate the most common pitfalls that have occurred in terrestrial and freshwater restoration projects. Lake (2001) outlined five common impediments to the development of restoration ecology, including (1) the focus on small projects due to the reluctance of resource managers to undertake significant projects, (2) poorly designed projects (3) the lack of appropriate monitoring of projects (4) a pervasive lack of reporting on progress and outcomes of projects and (5) issues associated with scaling up (temporally and spatially). Finally, the potential risk of engaging in restoration activities must be contrasted against the risk of doing nothing. The risk of decline in key habitats and their potential loss through the cumulative impacts of climate change and local pressures makes the imperative for more effective and efficient techniques urgent.

In this report we review the state of four critical marine and coastal environments in Australia; seagrass meadows, kelp forests, shellfish reefs, and coastal saltmarsh wetlands, and evaluate recent advances in restoration to improve conservation outcomes associated with MNES listed under the EPBC Act.

# 2 SEAGRASS MEADOWS

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## 2.1 Global role of seagrass meadows

Seagrasses are a polyphyletic group of marine flowering plants that are distributed globally, inhabiting coastal margins and estuaries on every continent, except Antarctica (Short *et al.* 2007). Although the diversity of seagrass species is not particularly high (~72 species globally, Short *et al.* 2011), the global success of this small group of marine flowering plants can be attributed to unique ecological, physiological, and morphological adaptations to a completely submerged existence (den Hartog 1970; Les *et al.* 1997; Phillips and Menez 1988). These adaptations include:

- an adaptation to survive in high, and in some cases varying, salinity
- an ability to grow whilst completely submerged by developing internal gas transport, epidermal chloroplasts and loss of stomata
- the use of an anchoring system to withstand water movement
- the development of submarine pollination strategies
- adaptations to enhance marine dispersal
- an ability to compete with other species in the marine environment

These adaptations have allowed them to flourish across a broad range of environmental settings (Short *et al.* 2007) including; hypo-saline to hyper-saline estuaries and bays; intertidal sand and reefs platforms down to ~80m depth; wave-exposed reefs and coarse sediments to sheltered environments with muddy and phyto-toxic sediments; as well as near-freezing temperatures via dormant seeds to near 40$^{o}$C within intertidal rock-pools.

In many locations, seagrasses form extensive ecosystems, often referred to as seagrass beds or seagrass meadows. They are considered to be one of the most important shallow-marine ecosystems to humans, being highly productive, and providing nutrient and resource linkages to other high value coastal ecosystems, including coral reefs, mangrove forests, and open ocean ecosystems (Beck *et al.* 2001; Heck *et al.* 2008). They also have an exceptional carbon sequestration capacity (Fourqurean *et al.* 2012) and an ability to stabilise sediments and attenuate wave energy, which can buffer coastlines and coastal structures from erosion (Orth et al, 2006). Seagrass meadows also help maintain water quality (Hemminga and Duarte 2000; Moore 2004), and provide food, habitat, and nursery grounds for a large diversity of ecologically and economically important fauna and flora (Beck *et al.* 2001).

There have been very few studies of the direct economic value of seagrasses. Costanza *et al.* (1997) calculated a global value of annual ecosystem services for seagrass of US$19,004 per hectare per year.

## 2.2 Global status of seagrass meadows

An extensive global survey of seagrass status has been completed in the recent past (Waycott *et al.* 2009) and this global effort will not be duplicated here. Instead, we will summarize the important outcomes from this review.

In the largest study of its kind, Waycott *et al.* (2009) analysed 215 studies of seagrass beds in shallow coastal waters from around the world. They found seagrass is being lost from east and west North America, the Caribbean, Mediterranean, Europe, parts of East Asia, Southeast Asia, as well as tropical and temperate Australasia.

Nearly 30% of global seagrass beds have been lost since records began, and the rate of loss is accelerating. Since 1980, 29% of seagrass has disappeared and the overall rate of loss has accelerated from 0.9% a year, before 1940, to 7% a year, between 1990 and 2006. That is, every year about 110 square kilometres of seagrasses are being lost globally or one football field sized area every 30 minutes (Waycott *et al.* 2009).

Nutrients in sewage and run-off from agriculture and industry are the major cause of seagrass death (Waycott *et al.* 2009). Nutrients trigger the growth of algae, plants and animals that grow above or on seagrass, and stop it from getting the sunlight it needs. For example, in Cockburn Sound, 80% of seagrasses (or 1200 hectares) have been lost over the last four decades and can be tied directly to nutrient input in the form of nitrogen (Cambridge *et al.* 1986).

The global threats to seagrasses have received considerable attention from a number of authors and their efforts have not been duplicated here (Orth *et al.* 2006; Waycott *et al.* 2009). Instead we provide a short summary of known natural and human induced threats to seagrass ecosystems (Table 2.1). In many cases it seems likely that seagrass areas have declined as a result of a combination of threats.

Table 2.1: Summary of common natural and human induced threats to seagrass ecosystems

| | Description |
|---|---|
| *Natural threats* | |
| Biological | Grazing by sea urchins, sirenians, geese, or removal by foraging rays |
| Meteorological | Storms, cyclones, hurricanes and wave action |
| *Human threats* | |
| Dredging | Capital or maintenance dredging works |
| Trawling | Various benthic trawling devices used in the fishing industry |

| | |
|---|---|
| Deliberate clearance | For example to clean tourist beaches |
| Erosion from altered hydrological regimes | Coastal development and the building of sea defences has significant effects on the flow of currents in nearshore waters |
| Anchor and propeller scar damage | Recreational & commercial fisherman |
| Land reclamation | |
| Aquaculture | Direct smothering from fish food and faeces, indirect algal growth from increased nutrients in the water column |
| Sedimentation | Higher turbidity reduces light levels. Very high sedimentation smothers entire seagrass beds |
| Pollution | Can have toxic or eutrophic effects. With high levels of increased nutrients, photosynthesis can be reduced by excess epiphytic overgrowth, planktonic blooms or competition from macroalgae |
| Climate change | Potential threats from rising sea levels, localised decreases in salinity, damage from UV radiation, and unpredictable impacts from changes in distribution and intensity of extreme events. Possible increases in productivity resulting from higher $CO_2$ concentrations |

## 2.3 Success and failure of seagrass meadow restoration around the world

There have been several restoration reviews in recent years that outline seagrass restoration attempts and their successes or failures (Fonseca et al, 1998; Paling et al 2009; van Katwijk *et al.* 2009; Statton et al 2012; van Katwijk et al 2016). These reviews span almost two decades of restoration research and practice, and share the following trends and outcomes:

- Despite an estimated loss of 30% of seagrass worldwide (Waycott *et al.* 2009), restoration programs as a means of recovering lost seagrass ecosystems have been largely unsuccessful (Fonseca *et al.* 1998; Paling *et al.* 2009; van Katwijk *et al.* 2009; Statton et al 2012; van Katwijk et al 2016)
- There have been some highly successful, large-scale restoration programs (e.g. Orth *et al.* 2012), however, despite decades of restoration research and practice relatively little global progress has been achieved (van Katwijk *et al.* 2016)
- Although there are large losses in almost every bioregion (Waycott et al 2009), most restoration efforts outside of Australia have taken place in the US and Europe and more recently in parts of Asia
- Regarding planting procedures, the most important factors affecting the success of revegetation trials were anchoring technique and plant material. During the first months after planting, any anchoring of rhizome fragments or seedlings enhanced survival in comparison to no anchoring. Restoration efforts have trialled a variety of planting units including transplanting individual rhizomes, small cores, larger (> 1m) sods as well as seeds and seedlings. Seedlings consistently perform worse than rhizome fragments. In contrast, the most successful large-scale reintroduction to date

was through annual re-seeding of *Zostera marina* in the US which spanned a decade (Orth *et al.* 2012)
- The scale of restoration sites have typically been small, less than 10 hectares (Statton *et al.* 2012; van Katwijk *et al.* 2016; but see Orth *et al.* 2012) but trends from a recent meta-analysis indicate that successful regrowth of the foundation seagrass species appears to require crossing a minimum threshold of reintroduced individuals (van Katwijk *et al.* 2016). That is, there is a requirement of a critical mass for recovery, which may also hold for other foundation species showing strong positive feedback to a dynamic environment
- Success of restoration efforts depends on a variety of factors such as scale of the restoration, spreading the risk across multiple locations or plots, site selection, environmental conditions and understanding environmental bottlenecks. Cost and clear objectives need to be set out in the planning stage (Fonseca *et al.* 1998; van Katwijk *et al.* 2009; Statton *et al.* 2012; van Katwijk *et al.* 2016).

## 2.4 Australian role of seagrass meadows

Australia is a biodiversity hotspot for seagrasses with a high level of endemism including some 16 species unique to Australian waters (Table 2). These seagrasses provide and perform a suite of critical ecosystem services.

### 2.4.1 Carbon stocks

Seagrass meadows in Australia are globally significant carbon sinks and support persistent carbon stocks. For example, Shark Bay holds one of the largest carbon stores of seagrasses globally (Fourqurean *et al.* 2012). In 2018, when this study was compiled, there were about 4,000 square kilometres (400,000 hectares) of seagrasses in the bay, which places it among the largest seagrass meadows that have been recorded in the world. Fourqurean *et al.* (2012) calculated the amount of carbon dioxide stored in the seagrass meadows in Shark Bay was 350 million tonnes of carbon (calculated by multiplying the average carbon per hectare, 884 tonnes, by 400,000 hectares of seagrass).

### 2.4.2 Stabilisation

The extensive root and rhizome system in seagrasses, which extends both vertically and horizontally, helps stabilise the sea floor in a manner similar to the way coastal plants prevent sand dune erosion. Sea floor areas that are devoid of seagrass are vulnerable to intense wave action from currents and storms. For example, heavy losses of 6200 ha of seagrass off the Adelaide metropolitan coast since 1949 have had substantial implications for beach management, fisheries and biodiversity (Tanner *et al.* 2014). Because of this seagrass loss, there have been substantial changes to the coastal ecosystem of Adelaide. Originally, seagrasses extended into shallow waters (~2 m depth; Fox *et al.* 2007) and stabilised coastal sediments (Fotheringham 2002). In part, as a consequence of seagrass loss, some 100 000 $m^3$ of sand is now deposited on Adelaide's beaches each year (Fox *et al.* 2007), while at the same time, increased longshore movement of sand has contributed to beach erosion and the need for an ongoing sand management programme costing $5 million per annum.

### 2.4.3 Associated flora and fauna

A vast array of species are found within seagrass ecosystems, and many are obligate members of the seagrass ecosystem that are found nowhere else. For example, 66 species of macro-algal epiphytes were recorded on the seagrass *Amphibolis antarctica* from 34 locations in Shark Bay, Western Australia (Kendrick *et al.* 1988). Fifty percent of the species are endemic to southern temperate Australia (Kendrick *et al.* 1988).

Some species utilise seagrass meadows for certain components of their life history, using them as breeding or nursery areas, or settling there as adults. Many more species are found across a broad range of marine habitats, but regularly inhabit seagrass areas. While we are still a long way from developing an estimate of total species numbers within seagrass ecosystems, and even further from establishing which of these are wholly dependent on these systems, the totals may be very large indeed. In addition, the close association of seagrass ecosystems with coral reefs and mangrove forests will greatly boost the numbers of facultative inhabitants of these ecosystems in these areas, and it seems like that the total figures will number tens, perhaps hundreds of thousands of species. Looking at a broader faunal list, Hutchings (1994) listed some 248 arthropods, 197 molluscs, 171 polychaetes, and 15 echinoderm species from Jervis Bay in New South Wales, Australia. While these estimates are lower than many coral reef biodiversity statistics (for example 1500 fish on the Great Barrier Reef; Spalding *et al.* 2001), these estimates still highlight the importance of seagrass meadows as an often overlooked source of biodiversity.

### 2.4.4 Nursery areas and habitat

The relative safety of seagrass meadows provides an ideal environment for juvenile fish and invertebrates to conceal themselves from predators (Heck *et al.* 1997; Butler and Jernakoff 2000). Many of Australia’s recreationally and commercially important marine life from coastal and estuarine ecosystems can be found in seagrass meadows during at least one early life stage (Butler and Jernakoff 2000). While seagrasses are ideal for juvenile and small adult fish to escape from larger predators, many in-faunal organisms (animals living in soft bottom sediments) also live within seagrass meadows. Many taxa such as clams, worms, crabs, starfishes, sea cucumbers, and sea urchins, use the buffering capabilities of seagrasses to provide a refuge from strong currents (Heck *et al.* 1997). The dense network of roots established by seagrasses also helps deter predators from digging through the substratum to find in-faunal prey organisms. Seagrass leaves provide a place of anchor for seaweeds and for filter-feeding animals like bryozoans, sponges, and forams.

### 2.4.5 Water Quality

Seagrasses help trap fine sediments and particles that are suspended in the water column, which increases water clarity (Moore 2004). When a sea floor area lacks seagrass communities, the sediments are more frequently stirred by wind and waves, decreasing water clarity, affecting marine animal behaviour, and generally decreasing the recreational quality of coastal areas (Moore 2004). Seagrasses also work to filter nutrients (Moore 2004) that may arise from land-based industrial discharge and stormwater runoff before these nutrients are washed out to sea and to other sensitive habitats such as coral reefs.

Table 2.2: List of single country endemic species of seagrasses

| Countries | Species |
|---|---|
| Australia | *Amphibolis antarctica, Amphibolis griffithii, Cymodocea angustata, Halophila australis, Halophila capricorni, Halophila tricostata, Posidonia angutstifolia, Posidonia australis, Posidonia coriacea, Posidonia denhartogii, Posidonia kirkmanii, Posidonia ostenfeldii, Posidonia robertsonae, Posidonia sinuosa, Thalassodendron pachyrhizum. Zostera mucronata* |
| Bermuda | *Halodule bermundensis* |
| Brazil | *Halodule emarginata* |
| Japan | *Phyllospadix japonicus, Zostera caespitosa* |
| Madagascar | *Halophila stipulacea* |
| New Zealand | *Zostera novazelandica* |
| USA | *Halophila hawaiiana, Halophila johnsonii* |

## 2.5 Australian status of seagrass meadows

Australia has an estimated 51 000 km$^2$ of seagrass meadows within its waters. Losses of seagrasses as a result of natural and human induced perturbations have been reported from across Australia in a recent book chapter (Statton *et al.* 2018; Appendix I) with the majority occurring along the heavily populated eastern region of Queensland and New South Wales, and mid-western region of Western Australia. Subsequently, Australia has recorded a 5.5% loss of seagrasses since the 1930's. Whilst these estimates only represent those losses at specific sites that have been observed and/or reported, they do not consider seagrass recovery or seasonal variability at any of these reported sites, therefore could still be a conservative representation of overall losses.

The largest losses (in terms of rate of loss) were recorded from the tropics and subtropics. In 1985 Cyclone Sandy destroyed over 18 300 ha of seagrass from West Island to Limmen Bight, Northern Territory (Poiner *et al.*, 1987), although 10 years later the meadows had largely recovered. In tropical areas, transitory meadows of opportunistic and colonising seagrass species dominate (Kilminster *et al.*, 2015). These colonising species have faster growth rates and large numbers of seeds in the seed bank, and can therefore recover faster following disturbance events (Rasheed *et al.*, 2014). In contrast, persistent species in more temperate zones have comparatively slower recovery rates (Walker and McComb 1992; Irving *et al.* 2010). In the subtropical region of Shark Bay, Western Australia, where there is considerable overlap of tropical and temperate seagrass species (Walker *et al.* 1988; Kendrick *et al.* 2012), an abnormal marine heat wave event (summer of 2011) combined with extreme cyclonic flooding (Fraser *et al.* 2014; Thomson *et al.* 2015) caused up to 86 000 ha loss of the temperate seagrass *Amphibolis antarctica* (Arias-Ortiz *et al.* 2018). In temperate zones, seagrass losses are generally a result of human activities, rather than natural events such as cyclones.

## 2.6 Seagrass meadows and Matters of National Environmental Significance

Seagrasses are a prominent feature of tropical and temperate coastlines of Australia and are a vital habitat relating to MNES. MNES (see Table 2.3) include World Heritage Listed Areas (e.g. Shark Bay, Great Barrier Reef), Ramsar wetlands (e.g. The Coorong, Pittwater, Western Port, Roebuck Bay), Commonwealth Marine Parks (e.g. Geographe Bay, Great Barrier Reef) and ecological communities listed as Critically Endangered, Endangered or Vulnerable under the EPBC Act (e.g. *Posidonia australis* seagrass habitat in NSW).

For many MNES relating to seagrass habitat, there have been significant losses of seagrass habitat with several local and regional drivers of impact (see Table 2.3). While there have been several publications detailing the specific status of these MNES habitats and drivers of loss (Table 2.3), for the purposes of this report we aim to provide some detail on a single MNES, Shark Bay World Heritage Area. Shark Bay World Heritage Area has received little attention despite recent climatic events and ongoing human use potentially altering the natural values for which this location was granted world heritage status.

Shark Bay is a World Heritage Area ecosystem supporting extensive areas of seagrass (4000km$^2$; Walker *et al.* 1988). Shark Bay is remote, situated 800 km from the nearest major city and with a permanent population of less than a 1000 people. Despite its remoteness, a range of anthropogenic pressures are present. In tourism season, the population of Shark bay increases 10-fold (from 10,000 to over 100,000). Further, although Shark Bay is World Heritage listed, this does not preclude coastal development. There are minor industrial, port and aquaculture developments, significant fishing and boating activity, boat moorings and tourism activities.

Shark Bay is a large (13 000 km$^2$) shallow (mainly 10 m) subtropical embayment, partly separated from the Indian Ocean by Pleistocene dunes (Logan and Cebulski 1970). Water circulation with the open ocean has been reduced further by a series of ridges and sills created from biogenic calcareous sediment deposition (Logan and Cebulski 1970). The sills in Shark Bay restrict water circulation, which, in combination with high evaporation rates, has contributed to the hypersalinity gradient. The hypersalinity gradient in the bay (salinity 35–70 ppt) is also permanent because of annual evaporation rates exceeding rainfall (Logan *et al.* 1970).

Shark Bay is dominated by calcareous sediments, which adsorb available phosphorus (Short 1987), decreasing phosphorus concentrations enough to limit the growth of aquatic biota (Smith 1984; Atkinson 1987). Despite this unique biogeochemistry, the bay has a high diversity of seagrass species, supporting 12 species (Walker *et al.* 1988), a substantial proportion of the total global species diversity of 72 species (Short *et al.* 2011). Shark Bay is dominated (biomass) by temperate Australian seagrass species (*Amphibolis antarctica*, *Posidonia australis*) growing at the northern extent of their geographic range.

More than 860 km$^2$ of seagrass meadows have been degraded within Shark Bay (Arias-Ortiz et al 2018), with minimal natural recovery even after a prolonged period (7 years). An abnormal marine heat wave event (summer of 2011) combined with extreme cyclonic flooding (Fraser *et al.* 2014; Thomson *et al.* 2015) caused the extensive loss or degradation of the temperate seagrass *Amphibolis antarctica*. Furthermore, *Posidonia australis* experienced reproductive failure in the few sites that were monitored within Shark Bay (Sinclair *et al.* 2015) although over the following years, larger spatial surveys have revealed other locations within Shark Bay with and without reproductively viable *P. australis* populations (Statton, unpublished data).

Table 2.3: Status and drivers of loss of seagrass habitat (by state) relating to MNES

| Timeframe | Location | Area of loss (ha)* | Species impacted | Drivers of loss | MNES | Reference |
|---|---|---|---|---|---|---|
| **New South Wales** | | | | | | |
| 1930-1999 | Gunnamata Bay | 16 | *Posidonia australis, Zostera muelleri* | Severe storms; Bait digging; | IUCN red list (*Posidonia australis*) | Williams and Meehan 2001 |
| 1942-1999 | Burraneer Bay | 5 | *Posidonia australis, Zostera muelleri* | Dredge disposal | IUCN red list (*Posidonia australis*) | Williams and Meehan 2001 |
| 1942-1999 | Cabbage Tree Basin | 12 | *Posidonia australis, Zostera muelleri, Halophila sp.* | Sand migration | IUCN red list (*Posidonia australis*) | Williams and Meehan 2001 |
| 1948-1994 | Merimbula Lake | 47 | *Posidonia australis* | - | IUCN red list (*Posidonia australis*) | Meehan 1997 |
| 1951-1999 | Lilli Pilli Point | 7 | *Posidonia australis, Zostera muelleri* | Channel Dredging and sand migration | IUCN red list (*Posidonia australis*) | Williams and Meehan 2001 |
| 1957-1994 | Wagonga Inlet | 8 | *Posidonia australis* | - | IUCN red list (*Posidonia australis*) | Meehan and West 2002 |
| 1957-1998 | Bermagui River | 14 | *Posidonia australis* | - | IUCN red list (*Posidonia australis*) | Meehan and West 2002 |
| 1961-1998 | St Georges Basin | 86 | *Posidonia australis* | - | IUCN red list (*Posidonia australis*) | Meehan and West 2002 |
| 1988-1995 | Wallis Lake | 518 | *Posidonia australis* | - | IUCN red list (*Posidonia australis*) | Dekker *et al.* 2005 |
| ____-1986 | Lake Macquarie | 700 | *Zostera capricorni**, Halophila ovalis, Ruppia megacarpa* | Light reduction (eutrophication) | IUCN red list (*Posidonia australis*) | King and Hodgson 1986 |
| ____-1986 | Tuggerah Lakes | 1,300 | *Zostera capricorni,**, Halophila ovalis, Ruppia megacarpa* | Light reduction (eutrophication) | IUCN red list (*Posidonia australis*) | King and Hodgson 1986 |
| **Queensland** | | | | | | |
| 1992-1993 | Hervey Bay | 100,000 | *Zostera capricorni*** | Flooding; Cyclone | Great Barrier Reef World Heritage List, Commonwealth Marine Park | Preen *et al.* 1995 |
| 1995-2012 | Lizard Island | 8 | *Thalassia hemprichii, Halodule uninervis* | Eutrophication | Great Barrier Reef World Heritage List, Commonwealth Marine Park | Saunders *et al.* 2015 |

| Timeframe | Location | Area of loss (ha)* | Species impacted | Drivers of loss | MNES | Reference |
|---|---|---|---|---|---|---|
| 1998-2001 | Urangan | 91 | *Zostera capricorni*** | - | Great Barrier Reef World Heritage List, Commonwealth Marine Park | Campbell and McKenzie 2004 |
| 1998-1999 | Wanggoolba Creek | 120 | *Zostera capricorni*** | - | Great Barrier Reef World Heritage List, Commonwealth Marine Park | Campbell and McKenzie 2004 |
| 1998-2002 | Northern Great Sandy Strait | 1,896 | *Zostera capricorni*** | - | Great Barrier Reef World Heritage List, Commonwealth Marine Park | Campbell and McKenzie 2004 |
| 2001-2013 | Townsville | 3,700 | *Zostera muelleri, Halophila ovalis, Halophila spinulosa* | Cyclone | Great Barrier Reef World Heritage List, Commonwealth Marine Park | Coles *et al.* 2015 |
| 2002-2013 | Gladstone | 1,600 | *Zostera muelleri, Halophila ovalis, Halodule uninervis* | Flooding, dredging, land reclamation | Great Barrier Reef World Heritage List, Commonwealth Marine Park | Coles *et al.* 2015 |
| Great Barrier Reef World Heritage List, Commonwealth Marine Park | Hay Point | | *Halophila decipiens* | Dredging Change in rainfall patterns | Great Barrier Reef World Heritage List, Commonwealth Marine Park | York *et al.* 2015 |
| Great Barrier Reef World Heritage List, Commonwealth Marine Park | Cairns | 700 | *Zostera muelleri, Halophila ovalis* | - | Great Barrier Reef World Heritage List, Commonwealth Marine Park | Coles *et al.* 2015 |
| ____-1990 | Moreton Bay | 257 | *Zostera capricorni*** | Sediment burial | Great Barrier Reef World Heritage List, Commonwealth Marine Park | Kirkman 1978 |
| **South Australia** | | | | | | |
| 2000-2010 | The Coorong | Impacted, complete loss by 2010 | *Ruppia tuberosa* | Millenium Drought | Ramsar wetland | van Dijk pers. comm. |
| **Tasmania** | | | | | | |
| 1948-1990 | Pittwater | 1,201 | *Zostera muelleri, Heterozostera tasmanica, Halophila australis* | | Ramsar wetland | Hamdorf and Kirkman 1995 |

| Timeframe | Location | Area of loss (ha)* | Species impacted | Drivers of loss | MNES | Reference |
|---|---|---|---|---|---|---|
| **Victoria** | | | | | | |
| ____-1983 | Western Port | 17,800 | *Heterozostera tasmanica, Zostera muelleri* | Sedimentation of fine muds | Ramsar Wetland | Bulthuis 1983 |
| | Gippsland Lakes | Impact but extent unknown | *Zostera mulleri* | | Ramsar wetland | |
| | Corner Inlet | Impacted but extent unknown | *Posidonia australis, Zostera mulleri* | | Ramsar wetland | |
| **Western Australia** | | | | | | |
| 2011-2018 | Shark Bay | 86, 000 | *Amphibolis antarctica* | Thermal stress combined with light stress | Shark Bay, World Heritage List | Fraser *et al.* 2014; Thomson *et al.* 2015; AriasOrtiz et al 2018 |

## 2.7 Success and failure of seagrass meadow restoration in Australia

### 2.7.1 Habitat enhancement

For almost 50 years there have been active attempts to revegetate (ecological restoration) or understand how to revegetate (restoration ecology) seagrasses in Australia (Statton *et al.* 2018). A large proportion of these restoration projects (~70%, Appendix II) have had a research objective (e.g. testing ecological theories, techniques, or locations) rather than commercial-scale restoration attempts. This skew in objective will clearly have an effect when comparing the size of areas re-planted, scalability, duration of success and monitoring (discussed below and see van Katwijk *et al.* 2016). Here, for the purposes of defining the success or failure of a revegetation attempt or habitat enhancement (this section), we will base success on the most common measures of success recorded during seagrass revegetation attempts; survival, increase in shoot density and/or expansion of plants.

From the 1970’s until 2016, there have been up to 118 seagrass revegetation attempts across Australia (Appendix II). The number of revegetation attempts have been dis-proportional across the decades, with exponential increases in the number of projects up until the turn of the millennium. The 1970’s represent 3% of revegetation attempts, the 1980’s account for 21% of attempts, by far the greatest surge in attempts occurred in the 1990’s with almost 50% of revegetation trials within this decade, while the remaining 30% of revegetation attempts have been carried out in the 16 years since 2000, indicating a decrease (or lack of reporting) in the number of revegetation attempts from the 1990’s.

Despite decades of restoration practice across Australia on Australian seagrass species, our ability to improve the survivorship of plants still remains highly variable. There have been a number of highly successful re-plantings, with approximately 13% of studies showing greater than 90% of planting-units surviving or showing expansion of surviving plants to form meadows (e.g. Paling *et al.* 2001a; Bastyan and Cambridge 2008; Verduin *et al.* 2010; M. Waycott *pers. comm.*; Irving *et al.* 2010). However, there have been a far greater number of studies (43%) showing less than 10% survival, and almost 60% of studies showed less than 25% survival. The low and variable survivorship, which is common across the 46 years of seagrass revegetation in Australia, follows global trends.

### 2.7.2 Location and species

The majority of revegetation attempts (Appendix II) have been located in Western Australia (62), followed by South Australia (23), New South Wales (17), Queensland (13), Victoria (2) and Northern Territory (1). Across Australia, *Posidonia spp*. (*P. australis, P. sinuosa, P. coriaceae,* and *P. angustifolia*) have been involved in the greatest number of restoration attempts (52) with the majority occurring in Western Australia (39), then NSW (8) and South Australia (6). *Amphibolis spp.* (*A. antarctica* and *A. griffithii*) have been the next most popular restoration species with 36 revegetation attempts, spread across Western Australia (23) and South Australia (13). *Zostera spp*. (*Z. capricornii* and *Z. muelleri*) and *Heterozostera spp*. (*H. tasmanica* and *H. nigricaulis*) have been the most favoured species on the east coast of Australia with 20 revegetation attempts including NSW (9), Queensland (9) and Victoria (2). Other species such as *Thalassia hemprichii, Cymodocea spp., Halodule uninervis, Halophila spp., Syringodium isoetifolium* and *Ruppia sp.* have also been trialled, but rarely.

### 2.7.3 Environment

The majority of revegetation attempts (Appendix II) have taken place in marine environments (104) with far fewer in estuaries or inlets (14). The majority of programs focused on subtidal plantings (92) compared to intertidal plantings (18) with more than 70% of plantings reported in 2-10m water depth, 10% were planted at greater than 10m depth, whereas 20% were less than 2m. Revegetation attempts in the marine subtidal environment have typically been in high to high/medium wave exposed sites for species like *Amphibolis spp, P. coriacea, P. angustifolia* and *P. sinuosa* (Kirkman 1995; Kirkman 1999; Irving 2009; Irving *et al.* 2010; Wear *et al.* 2010; Irving *et al.* 2014; Tanner 2015). Replanting *Posidonia australis* has tended to occur in relatively sheltered embayments and estuaries (Meehan and West 2002; Bastyan and Cambridge 2008) to moderately exposed locations (Kirkman 1995; Kirkman 1999; Verduin et al 2010). For smaller, colonising species of seagrass such as *Cymodocea spp., Zostera spp.* and *Heterozostera spp.,* revegetation attempts have tended to take place in sheltered bays, estuaries and inlets (Gibbs 1997).

### 2.7.4 Techniques

Re-planting techniques have been a strong focus for much of the seagrass restoration research, with studies trialling different planting unit types and sizes, anchorage approaches, sediment stabilisation techniques, fertilisation, growth hormones and mechanical planting systems (Appendix II; Table 2.4). Numerous short-term pilot trials of seagrass transplanting methods have been undertaken in the Cockburn Sound area of Western Australia to develop improved survival of transplant units in the high wave-energy subtidal environment. Artificial seagrass mats have been trialled to stabilise sediment around transplant units. While the survival of *P. australis* transplants amongst these mats (up to 50% after 18 months in 60% of sites) was significantly greater than transplant units that were placed in bare sand, these mats did not prevent erosion and accretion around the transplant units (Campbell and Paling 2003). Similarly, van Keulen *et al.* (2003) trialled plastic mesh to stabilise sediments around transplanted plugs of *Posidonia sinuosa*, but transplant units did not survive beyond one year (van Kuelen et al 2003). Transplant spacing has been suggested to influence sediment stability and therefore restoration success, but in high energy environments, the spacing of *Posidonia coriacea* and *Amphibolis griffithii* transplant units did not influence sediment movement (Paling *et al.* 2003).

Trials in the late 1990's tested whether increasing transplant unit size would have a higher chance of success in the oceanic environment. Large sods of *Posidonia sp.,* and *Amphibolis griffithii* were transplanted to Success Bank at Cockburn Sound using underwater mechanical seagrass harvesting and planting machines (known as ECOSUB I and ECOSUB II). From 1996-1999, ECOSUB I was used to plant over 2000 0.25 $m^2$ sods into an approximate area of 3,000 $m^2$ of 25% seagrass cover (Paling *et al.* 2001a). The large sods seemed to provide sufficient anchorage in the high-energy environment and markedly improved transplant success in the area (Paling *et al.* 2001a). *Posidonia* species showed good survival rates two years after planting - 76.8% for *P. sinuosa* and 75.8% for *P. coriacea* (Paling *et al.*, 2001b). Seasonal timing of planting was also important for survival. Sods planted in spring or summer, were more likely to survive than those planted in autumn or winter (Paling *et al.*, 2001b). In early 2000, 280 0.55 $m^2$ seagrass sods were planted by the more efficient ECOSUB II (Paling *et al.* 2001a). These transplants showed comparable survival rates to those from ECOSUB I, and the restored area showed natural infilling by seagrass seedlings (Paling *et al.* 2001b).

Trial planting of both plugs and sprigs of *P. australis* and *P. sinuosa* at two sites in Cockburn Sound found that, although the individual site seemed to influence the survival of each species, the sprig method provided the highest growth rates of both species (Verduin *et al.* 2010). Manual transplantation of sprigs of *P. australis* and *P. sinuosa* was used for larger revegetation trials. After the first round of planting, survival of planted sprigs was low, 10% after a few months, because the twine used to tie the sprigs to the staple anchor degraded too quickly. Improvements to this technique resulted in higher survival rates after re-planting the following year combined with horizontal spread of established transplants resulted in one of the largest areas covered (~3 ha) for a *Posidonia sp.* restoration (Verduin *et al.* 2010).

In the sheltered estuarine waters of Oyster Harbour around Albany in Western Australia, high survival rates of manually transplanted and anchored *P. australis* sprigs have been reported (Bastyan and Cambridge 2008). Survival rates were 95% over six years from 1994, and 94% over four years from 1997. These sprigs, planted 1m apart and anchored with a wire peg, began to merge during the fourth year after transplanting, and by the end of the fifth year a complete seagrass bed with a plant density similar to adjoining natural seagrass beds (Basytan and Cambridge 2008). In contrast, in the nearby embayment of Princess Royal

Harbour, Albany, similarly *P. australis* transplanting trials were not as successful. Examination of rhizome expansion rates found *P. australis* non-apical sprigs extended far less than apical sprigs (Bastyan and Cambridge 2008).

Techniques to grow *P. australis* seedlings from seed in controlled tank conditions have been developed in Western Australia (Statton *et al.* 2012; Statton *et al.* 2013). Successful seedling rearing techniques have been developed for *P. australis* and these could result in an abundant supply of high quality seagrass propagules seedlings for out-planting in restoration programs.

To determine the ultimate use of cultured seedlings for seagrass restoration, some trial studies have investigated the survival and growth of seedlings planted in the 'wild'. In Western Australia, the planting of both laboratory-reared and natural seedlings of *P. australis* in sheltered natural waters around Albany between 2003 and 2006 found high initial short-term survival rates. After one year, the survival rate was 60% for seedlings raised from seed and 80% for those obtained from the natural environment (Oceanica Consulting, 2006). In early 2006, a trial planting of culture-reared seedlings amongst hessian bags was also conducted in Western Australia (Initial. Statton, *pers. comm.*). The survival of these planting units varied considerably and seemed dependent upon the quality of hessian bag used.

In South Australia, the use of culture-reared *Posidonia spp*. seedlings to form a large area of seagrass habitat was deemed impractical due to the difficulty in cultivating seedlings, the highly spatially and temporally variable sexual reproduction of local *Posidonia* species, and the slow growth rate of these species (Seddon *et al.* 2005). Research found that the growth of *Posidonia spp*. seedlings in culture is possible, but the survival rate of these seedlings was low due to excessive epiphyte growth and the level of shading over the tanks. It was suggested that cultured *Posidonia spp.* seedlings could be useful in accelerating natural succession in areas that are starting to be recolonised by fast-growing seagrass species (Seddon *et al.*, 2005).

Another method of seagrass restoration using *Posidonia spp*. seedlings is being trialled in South Australia. *Posidonia spp.* fruits collected from beaches are held in tanks until dehiscence (spontaneous opening at maturity) and the resultant seedlings are planted into sand-filled hessian bags that are then placed into the natural environment. The few seedlings that survived planting into the natural environment had very good growth rates over the longer-term (Wear, 2006).

Attempts at transplanting *Z. capricorni* have been conducted in NSW estuaries since 2000. These consisted of small-scale experiments to trial techniques for seagrass habitat restoration (Roberts *et al.* 2006). Individual shoot, multiple shoots and core methods were trialled to transplant *Z. capricorni*. Some of these transplanting attempts failed due to sediment movement or flood damage. However, a high percentage of survival (close to 100% after 12 months) was recorded when cores of *Z. capricorni* were transplanted into existing recipient beds of this species in Tuggerah Lakes. On the Gold Coast of Queensland, transplanting mixed and pure cores of *Z. capricorni* and *H. ovalis* into human-made depressions resulted in less than 50% survival after six months (McLennan and Sumpton, 2005).

In South Australia the success of the use of biodegradable hessian mats to stabilise sediments around cores and secure sprigs of mature *A. antarctica* and *H. tasmanica* transplants were unsuccessful (Seddon 2004). These methods were demonstrated to be unsuitable for forming large areas of seagrasses in the proposed area due to poor trial transplant survival rates, the relatively high effort and labour costs for these methods, and donor bed damage. It was suggested that this technique might only be suitable in low wave energy environments.

The comb-like grappling apparatus on *Amphibolis* seedlings can facilitate their attachment in a range of biodegradable hessian bags, strips or mats. While large numbers of seedlings can recruit onto hessian bags (157.2 seedlings $m^{-2}$); the retention of seedlings on these units declined, and after one year only 31.4% of these seedlings remained (Wear *et al.*, 2006). The method was found to be a non-destructive, cost-effective (i.e. costing $10,000 to rehabilitate one hectare of seagrass) method of *Amphibolis* seagrass restoration that could easily be deployed over large spatial scales (Tanner *et al.* 2014).

Table 2.4: Planting unit and method of planting

| Planting Unit | Method | Author |
|---|---|---|
| Sods | Mechanically Transplanted 0.5m$^2$ | Paling *et al.* 2001b |
| | | |
| Turfs | 20 tonne amphibious excavator | Lord *et al.* 1999 |
| | 21 tonne amphibious excavator | Lord *et al.* 1999 |
| | | |
| Plug | 5cm diameter | van Keulen *et al.* 2003 |
| | 10cm diameter | van Keulen *et al.* 2003 |
| | 15cm diameter | van Keulen *et al.* 2003; Western-Port Seagrass Partnership 2008 |
| | Hessian bags | Ganassin and Gibbs 2008; Tanner *et al.* 2014 |
| | Mechanical removal and transplant ECOSUB 1 & 2 | Paling *et al.* 2001a |
| | Meshed and unmeshed - sediment stabilisation | Lord *et al.* 1999 |
| | | |
| Sprig | 20-30 cm in length with 1-3 plagiotrophic shoots and 2-4 orthotrophic shoots | Meehan and West 2002 |
| | 20-25 cm lengths of rhizome (bare root) | Paling *et al.* 2007 |
| | 30 cm long | Irving *et al.* 2010 |
| | Anchored using staples | Lord *et al.* 1999 |
| | Anchored with pegs | Lord *et al.* 1999 |
| | Attached to plastic mesh and mesh anchored to sediment | Ganassin and Gibbs 2008 |
| | Fertilization with N & P | Hovey *et al.* 2012; Cambridge and Kendrick 2009 |
| | Growth hormones | Lord *et al.* 1999 |
| | | |
| Seedlings | Planted in hessian bags - 20 seedling per bag | Irving *et al.* 2010 |
| | PVC tube with intact sediment | Lord *et al.* 1999 |
| | Growool blocks or Jiffy pots | Kirkman 1999 |
| | Growool pots at different water depths | Kirkman 1999 |
| | Woven into rope on a grid | Lord *et al.* 1999 |
| | | |
| Seeds | Biodegradable substrates, mostly made of hessian (burlap), to enhance Amphibolis recruitment | Wear *et al.* 2010 |
| | Broadcast seeding from boat | Initial. Statton *pers. comm.* 2018 |
| | Broadcast seeding using divers | Statton *et al.* 2017a |
| | Transferring sediment seed bank | M. Waycott *pers. comm.* 2018 |

### 2.7.5 Duration

Monitoring time for 95% of revegetation attempts was typically three years or less across all species. Fifty percent of revegetation attempts were monitored for one year or less and only 5% of studies continued monitoring beyond three years. This appears to be related to the limited time frames or deadlines governing many grant-funding and commercial environmental-consulting activities (Statton *et al.* 2012). Clearly, revegetation attempts, which are also research programs, are skewing the results regarding the underwhelming success of revegetation attempts.

### 2.7.6 Scale

The majority of programs replanted into areas that were less than 100m$^2$ in area. Even when combining multiple plots the largest area attempted or fully restored was only a few hectares in area (BMT Oceanica Pty Ltd 2013). Therefore, successful, medium- to large-scale seagrass revegetation is rare, particularly compared to the 10's to 1000's of hectares lost in any one region where revegetation is required (Statton *et al.* 2018). The approaches used in the vast majority of these historical revegetation attempts have been such that the amount of effort, time and costs required to replant large areas would make restoration unfeasible. Indeed, in a recent review calculating the costs of coast restoration, Bayraktarov *et al.* (2016) estimated that seagrass restoration was the second most expensive ecosystem to restore (only superseded by coral reef restoration). In addition, seagrass projects reported the lowest median survival (38%) of the five ecosystems compared (coral reefs, seagrass, mangroves, salt-marshes, and oyster reefs). While transplanting adult plants will continue to play a role in seagrass restoration on small scales (e.g. seagrass salvage operations or in locations where natural seed availability is low), the future for large scale seagrass restoration will be through the use of seeds (see below). The advantages to using seeds for restoration are significant savings in time, effort and costs required to collect adult plant material, less damage to existing meadows, and potentially higher levels of genetic variation (outcrossed seeds will have many different genotypes as compared with plant material potentially collected from a few clones).

## 2.8 Recent advances and new ideas for seagrass meadow restoration

### 2.8.1 Case study 1: Translocation of the *Ruppia tuberosa* seed bank in the Coorong

The ecological health of the Coorong, SA was devastated by a drought from 2006 to 2010. Decreased water levels and increased salinities in the Coorong South Lagoon resulted in the rapid decline of *Ruppia tuberosa*. *R. tuberosa* populations have not naturally returned on a large scale within the Coorong, SA, due to a severely exhausted seed bank.

Lake Cantara, in the Coorong National Park, has an established and healthy population of *R. tuberosa*, and is the donor site for the translocation project. *R. tuberosa* seeds are about 1mm in size, black and tear-dropped shaped, and can be found in lake bed sediments. At Lake Cantara, seeds are in high density in the top layer of sediment. The seed is collected when Lake Cantara is dry in late summer and early autumn. A small excavator is used to scrape off the top 15mm of sediment, containing the seeds. Track mats are used to reduce the impact of the excavator (Fig 2.1a). The seed is collected in strips, with even-width gaps to promote faster recovery of the *R. tuberosa* seed bank in Lake Cantara. The sediment is then collected and transported to translocation sites on the Coorong (Fig. 2.1b). Planting is carried out when mudflats around the edge of the Coorong South Lagoon are exposed (when water levels are low, Fig. 2.1c). Planting sites are chosen based on water level predictions, as *R. tuberosa* grows best in water depths between 30cm and 100cm. Planting involves lightly agitating the mudflat surface, scattering the seed sediment, and then pressing it into the soil (Fig. 2.1d). Deeper sections of mudflats can have shallow water cover even at planting time. For these sections, the seed sediment is scattered directly into the water and local wave action keeps it in place. A total of 280 (14,080 bags) and 450 tonnes (30,100 bags) of sediment were translocated in 2013 and 2014 respectively. Bags were translocated to Policeman Point and Woods Well in 2013 and to Fat Cattle Point, Jacks Point and Seagull Island in 2014. An estimated area of around 20 ha and 41 ha were treated during the two restoration years. The restoration efforts were successful in that *R. tuberosa* did recolonise the areas transplanted.

While the restoration has helped recovery in the South lagoon, the process of recovery has been slow, and up until 2016, water levels have not been high enough to successfully complete the reproductive cycle. In particular, seed and turion (a wintering bud which becomes detached and remains dormant at the bottom of the water) density remain low compared to historical values. Ongoing monitoring of the system will identify if there is increased *R. tuberosa* recovery in the Coorong over longer time frames.


Figure 2.1. Stages in the *R. tuberosa* translocation action 2014/2015, (a) harvesting seeds in sediments at Lake Cantara, (b) stores of sediments containing seeds, (c), placement of stored sediments on day of spreading and (d), spreading actions.

### 2.8.2 Case study 2: Facilitating natural seedling recruitment of *Amphibolis* spp. with artificial substrates

Since 1949, there has been a loss of 6,200 ha of seagrass from the Adelaide coast, primarily due to overgrowth by epiphytic algae that thrived as a result of anthropogenic nutrient inputs, and turbidity (Tanner *et al.* 2014). Initial restoration efforts focused on adapting techniques used elsewhere, namely transplantation and the laboratory production of seedlings, however, success was limited. Observations during these trials suggested that the use of hessian to facilitate natural recruitment of *Amphibolis* spp. seedlings may work (Fig. 2.2). Subsequent work in 2004 trialled a range of different deployment options, with a standard hessian sack filled with around 20kg of sand being selected for most subsequent work. These bags can simply be dropped off a boat, and do not require any further manipulation by divers, making it easy and cheap to deploy.

An issue with early trials was uncertainty around the best time of year to deploy bags to maximise recruitment. Anecdotal evidence suggested that late winter or early spring would be the best time. In 2007-2013, a concerted effort to identify the timing of reproduction and recruitment was made, with bimonthly deployments of bags and collection of adult plants at four sites. May to August was shown to be the best period for bag deployment to maximise recruitment success, and *Amphibolis* spp. structural characteristics (stem density and length) were similar to those in natural meadows five years after bag deployment. Inter-annual variation in recruitment was present, but relatively minor. Early deployments that had started to coalesce into larger patches in 2013 have now formed several larger patches where the locations of individual bags can no longer be distinguished.

One of the issues experienced has been the rapid deterioration of some batches of bags. To address this, a series of trials were undertaken with Flinders University to develop coatings that would increase their durability. While these trials showed some promising results, the logistics and costs associated with treating bags meant that this approach was not pursued further.

Not only do the bags provide a mechanism for the successful facilitation of *Amphibolis* spp. recruitment, but also the resultant patches appear to be providing a similar ecosystem function to natural seagrasses. Epifaunal species richness and abundance was similar to natural seagrasses one year after *Amphibolis* spp. recruitment. However, it took three years, the same time it took for seagrass structure recover, for the epifaunal assemblage structure to mimic that of natural seagrasses. Infaunal assemblages recovered within two years. Both *Zostera* and *Posidonia* seagrasses have recruited into patches of restored *Amphibolis*, and larger fauna such as syngnathids (seahorses, pipefish and seadragons) also utilise the restored habitat.

To extend the applicability of the technique to other seagrasses, trials were also conducted with *Posidonia sp*. Due to the different life-history strategy and morphology of the two genera, *Posidonia sp.* had to be planted into the bags as seedlings by divers, as they do not naturally recruit to them. Seedlings planted in 2012 survived and grew well over the subsequent four years, and have produced multiple shoots.

Overall, sand-filled hessian bags deployed at small-scales during winter are an effective means for rehabilitating patches of *Amphibolis* spp. with minimal intervention, provided that there is a nearby source of recruits. Small-scale patches now appear to be functioning the same as nearby natural meadows. The focus now is on a series of one-hectare scale trials, as well as examining how the handling of bags prior to deployment may affect their integrity.

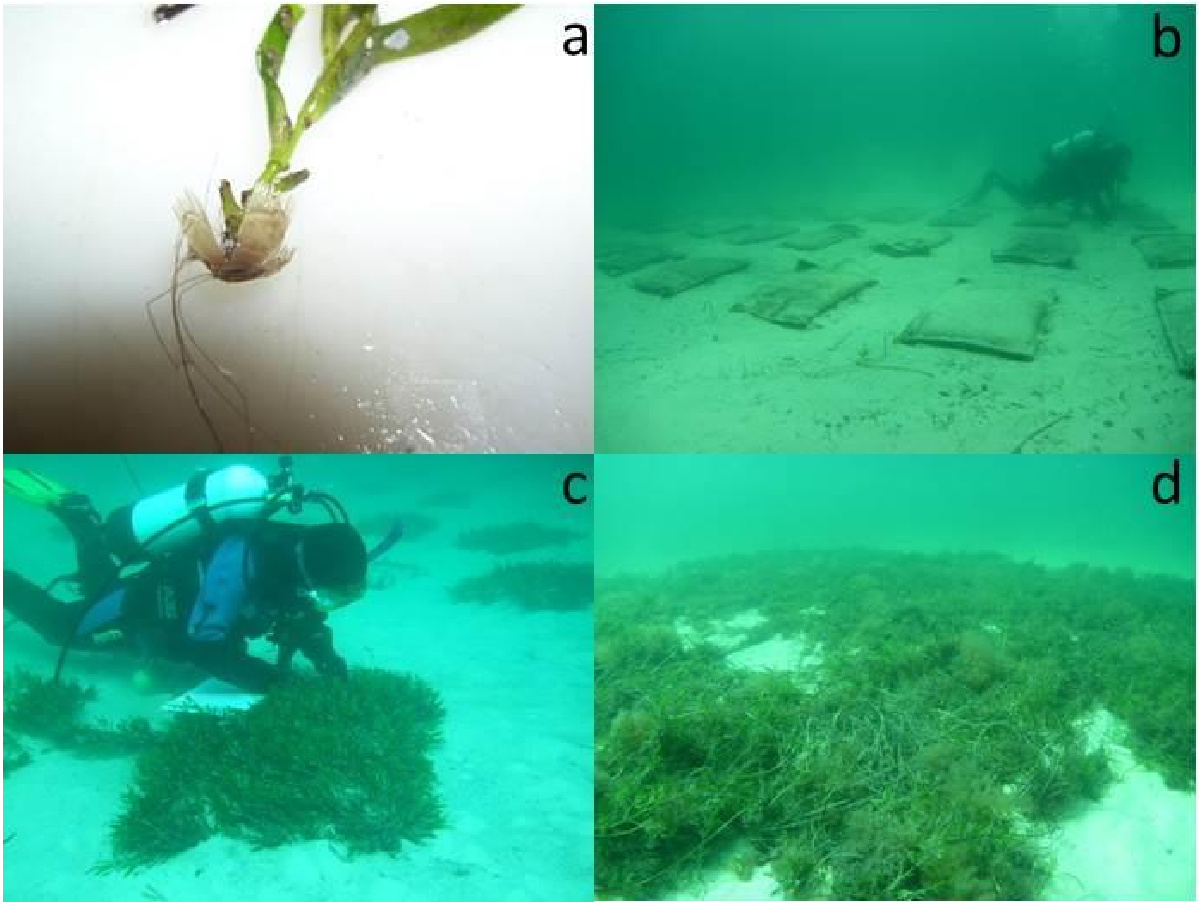


Figure 2.2: *Amphibolis* spp. recruitment facilitation approach showing; (a) *Amphibolis* spp. seedling with close-up of grappling hook to assist anchorage, (b) Recently deployed sand bags laid out for monitoring (c) 6-month old deployment covered in *Amphibolis* spp. seedlings (d) Restored *Amphibolis* spp. patch showing coalescence from ~40 bags.

### 2.8.3 Case study 3: Collection, processing and broadcast delivery of *Posidonia australis*

Cockburn Sound is a natural embayment approximately 16 km long and 7 km wide, to the west of the southern end of the Perth metropolitan area. Cockburn Sound has seen a 77% decline in seagrass cover (~2000 ha) since 1967, largely due to the effects of eutrophication, industrial development and sand mining (Kendrick *et al.* 2002). In small, localised areas, natural recruitment has been very successful, while other parts have not been able to recruit and recover naturally.

A number of techniques have been trialled in an attempt to develop efficient and cost-effective methods to regenerate seagrass meadows, including mechanically transplanting large sods, cores, transplanting sprigs and seedlings (Paling *et al.* 2001a,b; Verduin *et al.* 2010). Cost is a prohibitive factor for many of these methods, while availability of plant material and impact on existing meadows are prohibitive for others. The use of vegetative transplants has been the most widely used method, but restored areas are small, long-term success has not been good, and donor meadows can be negatively impacted. Many species of seagrass produce an abundance of seed (100’s-10 000’s $m^{-2}$) that offer a significant source of planting units, which like seed collection in terrestrial environments and unlike clonal material, can be obtained without direct negative impact on the donor population. The key objective in this research is the development of a large-scale collection, processing and remote seafloor delivery process for restoration of seagrasses from species with non-dormant, direct developing seed (*Posidonia australis*).

To address this objective we addressed the following more specific aims for this species by developing technologies to (1) collect fruit at maturity from source meadows (Fig. 2.3a) using purpose built nets, (2) process collected fruit in temperature controlled holding tanks by agitation via aeration (Fig. 2.3b) to obtain large quantities of seed material that settle on the bottom of the holding tank (Fig. 2.3c), and (3) trial approaches to effectively and efficiently deliver seeds to the restoration site which included; a) diver assisted, precision seeding by scattering seeds close to the sea-floor, and (b) remote, broadcast seeding from a boat. One of the major benefits of using the broadcast seeding method, as opposed to transplanting sprigs and shoots, is that seeds are negatively buoyant and naturally fall to the seafloor. Hence, there is no requirement for expensive and labour-intensive diving operations, especially when considering deeper sites or when there is low water visibility.

Pilot scale trials have shown good success. *P. australis* is seeded at densities of 200 seeds $m^{-2}$ into 3 x $25m^2$ replicate plots at four locations in Cockburn Sound. Seedling establishment success varies from 1% (2 seedlings $m^{-2}$) to 10% (20 seedlings $m^{-2}$) after 2 years. At 18 months, seedlings have begun to produce new shoots.

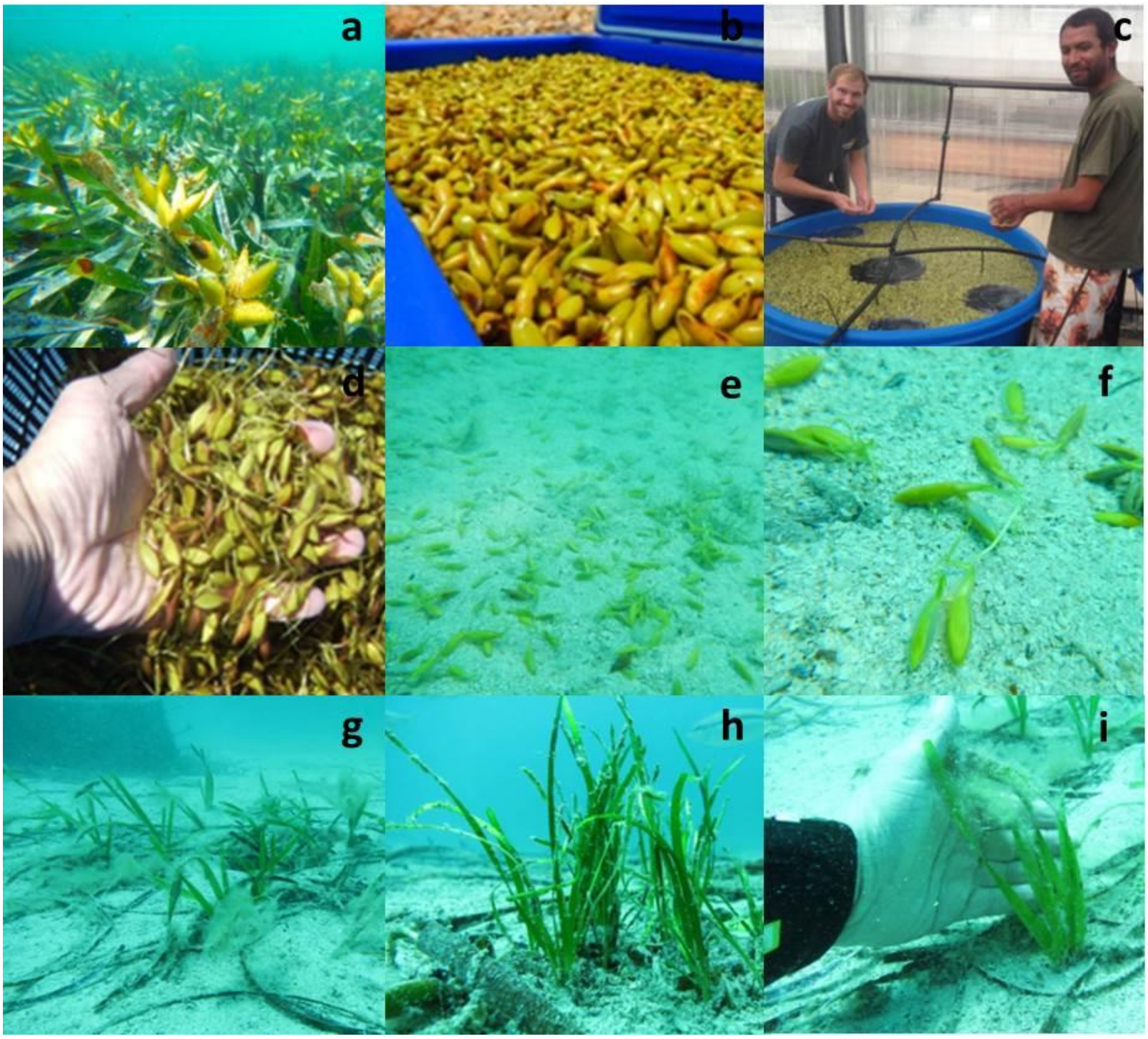


Figure 2.3: Mature *Posidonia australis* fruit prior to collection, (b), fruit in 100 litre cooler for transport back to lab, (c) processing fruit after collection, (d), after processing, seeds are clean and ready for delivery to field sites, (e), seeds scattered on surface of sediment (200 seeds $m^{-2}$), (f), close up of seeds settled on the sea floor, (g), 1 year old established seedlings, (h), seedlings established in high density, (i), two year old seedling with multiple shoots. Images by J. Statton.

### 2.8.4 Case study 4: Activating dormant *Halophila ovalis* seeds to stabilise dredge slopes

As a recently dredged area settles, currents and water movement wash loose sediment into the dredged channel. This creates the need for routine dredge maintenance which is both expensive and environmentally damaging. The presence of submerged aquatic vegetation on dredge slopes could naturally accelerate the stabilization of sediment and potentially reduce the need for continuous maintenance, thus saving money and minimizing environmental impact.

*Halophila ovalis* (Fig. 2.4) is a species of seagrass with ideal characteristics for restoration projects and revegetating dredge slopes in particular. *H. ovalis* is quick to grow and colonize, meaning it forms groundcover rapidly and would accelerate sediment stabilisation. It is also found at a range of depths, which suggests it could grow across a dredge slope despite varying light conditions. The presence of a seed dormancy period favours population persistence and also allows seeds to be transported and stored for future use.

This project will adopt the use of pre-treated seeds from successful terrestrial restoration techniques in order to 'activate' seeds prior to transferring to field sites and ultimately enhance germination success. The key objective in this research is to streamline seeding restoration practices by pre-treating *H. ovalis* seeds to optimize germination success and yield 'restoration-ready' seeds irrespective of destination site conditions. To achieve this objective, we are pre-treating seeds with different light quality and temperature treatments (Strydom *et al.* 2017; Statton *et al.* 2018). These recent studies of the effects of light quality and temperature on *H. ovalis* suggest that pre-treating seeds with red light wavelengths and a stepwise temperature increase from 15-25 °C simulating winter cold stratification will cue germination regardless of season and environmental conditions.

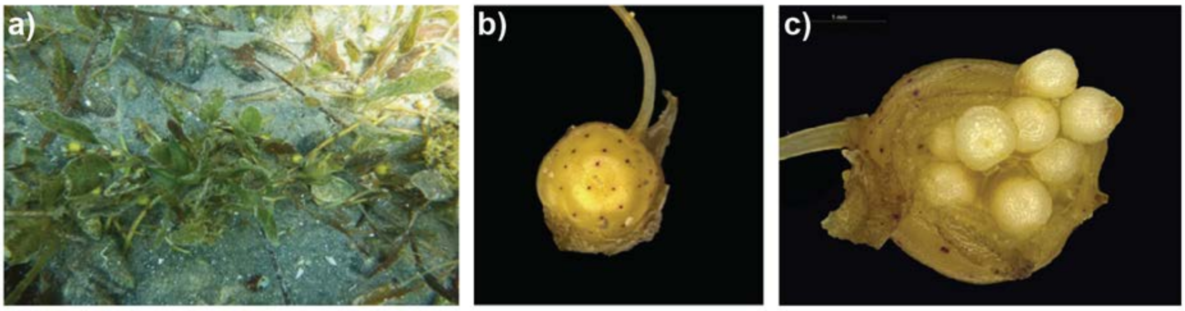


Figure 2.4: a) Mature *Halophila ovalis* fruit still attached to base of shoot on parent plant prior to collection, (b), *H. ovalis* fruit, (c), seeds being released from fruit.

## 2.9 Matters of National Environmental Significance that could benefit from on-ground restoration investment

1. *Shark Bay, Western Australia*
   The seagrass habitat relating to MNES that would show the greatest benefit for on-ground restoration investment and the easiest to tackle right now would be Shark Bay World Heritage Area, West Australia. Shark Bay has lost more than 860km$^2$ of *A. antarctica* habitat with 20% of the remaining meadows (*A. antarctica* and *Posidonia australis*) showing significant degradation from the marine heatwave in 2011 (Arias-Ortiz *et al.* 2018). We have already begun developing methodologies to revegetate seagrasses within Shark Bay, which have been quite successful, and would require only a scaling up of efforts (Statton and Kendrick 2018).

2. *Coorong wetland, South Australia*
   The second MNES that would benefit from on-ground investment would be the Coorong, SA, a Ramsar wetland. The millennium drought of 2000-2010 impacted water flows to the Coorong, causing the decline and subsequent loss from the system. There has been good success in regenerating *R. tuberosa* meadows within the Coorong via spreading topsoil containing the *Ruppia* seed bank. However, long-term success has

been unattainable. The main driver of poor success has been the variability in the flows from the Murray River into the Coorong. There will need to be significant investment in developing strategies to improve river flows before restoration will be successful.

3. *P. australis in New South Wales*
The third MNES that could benefit from restoration investment would be the *P. australis* communities on the NSW coast and estuaries. However, these endangered seagrass meadows suffer from problems arising from finding enough plant material to carry out a restoration. *P. australis* is a protected species within NSW and therefore removal of plant material for restoration is heavily regulated and/or prohibited. While programs are now being developed to overcome these issues (e.g. collecting plants as storm drift or using seeds collected as beach wrack) these are still in the developmental stages and have not yet proven to yield a suitable amount of viable plant material.

4. *Gladstone Harbour, Queensland*
A final MNES seagrass habitat that would benefit from restoration investment would be Gladstone Harbour, within the Great Barrier Reef World Heritage Area, Queensland. While there has been significant investment in understanding and mitigating the drivers of loss in this system, the collection, processing and delivery of plant material (i.e. seeds) is still in its infancy and has yet to be proven in the field.

## 2.10 Other benefits from seagrass meadow restoration

Presuming that restored seagrass meadows would have similar ecosystem services as natural meadows the benefits would be similar to those described in Section 2.4. Some of the other benefits stemming from restoration would include:

- the provision of nursery habitats for commercial fish species (McArthur and Boland 2006; Bertelli and Unsworth 2014)
- carbon sequestration (Macreadie *et al.* 2014; Marbà *et al.* 2015)
- protection of the coast against erosion (Fonseca and Cahalan 1992)
- water purification and nutrient cycling (Barbier *et al.* 2011 and Cullen-Unsworth *et al.* 2014)
- tourism and recreation (Cullen-Unsworth *et al.* 2014)
- developing a restoration economy, generating local/regional employment opportunities
- collaborations with the local indigenous Sea-Rangers, who are custodians of the Indigenous Protected Areas (IPA's), would add enormous value to the research and practice of restoration
- The exchange of knowledge with the rangers during projects would improve our collective knowledge of the importance of seagrass to food sources such as Green Turtles and Dugongs and as a habitat for other important food sources
- engaging with Traditional Owners should be considered as a Key Performance Indicator for any restoration program that overlaps with IPA's

# 3 RESTORING KELP HABITAT IN AUSTRALIA

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## 3.1 Global role of kelp forests

Kelp[1] dominate rocky coastal environments in temperate and subpolar latitudes around the world (Steneck *et al.* 2002; Smale *et al.* 2013; Schiel & Foster 2015; Krumhansl *et al.* 2016). These large, brown, habitat-forming seaweeds occur in intertidal and subtidal habitats, and range in size from less than a metre to over 40 m in length. Much like terrestrial forests, kelp forests are complex habitats with altered sub-canopy conditions (e.g. reduced light and water flow, Jackson and Winant 1983; Wernberg *et al.* 2005), and support entire communities of associated flora and fauna (Steneck *et al.* 2002; Teagle *et al.* 2017; Miller *et al.* 2018a). In addition to providing habitat, kelp also act as the trophic foundation of coastal food-webs by providing food for a broad suite of grazers, detritivores and microbes (Dayton 1985; Steneck *et al.* 2002; Graham *et al.* 2007). These trophic effects can reach even to deep waters and beyond the continental shelf (Thompson *et al.* 2011; Krumhansl and Scheibling 2012; Krause-Jensen and Duarte 2016; Filbee-Dexter *et al.* 2018).

As the foundation species of rocky reef ecosystems, kelp forests underpin high levels of coastal biodiversity and productivity (Smale *et al.* 2013; Schiel and Foster 2015; Bennett *et al.* 2016). In fact, kelp forests are some of the most productive habitats on earth, rivalling even the most intensively managed agricultural systems (Mann 1973). Kelp forests in Australia and elsewhere also support species of significant conservation (e.g. weedy seadragons, otters) and commercial (e.g. lobsters, abalone) importance (Steneck *et al.* 2002; Smale *et al.* 2013; Bennett *et al.* 2016). These underwater forests can also help ameliorate storm swells and coastal erosion by modifying local hydrography (Jackson and Winant 1983;

[1] The taxonomic definition of kelp is the large, brown seaweeds belonging to the Order Laminariales. However, some authors (e.g. Steneck & Johnson 2014) argue for a broader functional definition of kelp to include other large, brown canopy-forming seaweeds such as those from the Order Fucales (see also Schiel & Foster 2006). We adopt this broader functional definition of kelp, especially in light of the important contribution of fucalean seaweeds to coastal habitats in Australia (see Coleman & Wernberg 2017).

Løvås & Tørum 2001; Gaylord *et al.* 2007), while fast growth rates (up to 400 mm a day in some species; Schiel & Foster 2015) and high biomass also give them vital roles in cycling and sequestering coastal nutrients and carbon (Costanza, *et al.* 1997; Smale *et al.* 2013; Bennett *et al.* 2016; Krause-Jensen *et al.* 2018). Overall, kelp forests around the globe provide billions of dollars' worth of ecosystem services annually, and play a critical role in the health, function and productivity of coastal ecosystems.

## 3.2 Global status of kelp forests

Assessments of the global status of kelp forests are hampered by a lack of data and high geographic variation at local scales. The most comprehensive and recent analysis of the global status of kelp forests (Krumhansl *et al.* 2016) utilised data from 34 of the 99 global 'ecoregions' where kelp exists and found that 38% of studied ecoregions experienced declines in kelp over the past several decades. These regions of kelp decline include the North Sea, North-Central California, Central Chile, South Australia, the Bassian ecoregion (which covers Tasmania, Bass Strait and parts of Victoria) and the Manning-Hawkesbury ecoregion (covering central and northern New South Wales). In contrast, 27% of the studied ecoregions illustrated increases in kelp over the same period, including the South European Atlantic Shelf and Southern California. The remaining 35% of studied regions demonstrated no significant changes in kelp. Where kelp are declining, agents of decline include ocean warming, increasing abundance of herbivores (e.g. sea urchins) leading to destructive grazing, pollution, urbanisation, and invasive species (Airoldi & Beck 2011; Smale et al. 2013; Steneck & Johnson 2014; Ling et al. 2015; Krumhansl et al. 2016; Filbee-Dexter & Wernberg 2018). While the work of Krumhansl et al. (2016) represents the most comprehensive global analysis to date, it is worth noting that several of the areas showing increases in kelp abundance were those with relatively shorter time series of data.

## 3.3 Success and failure of kelp forest restoration around the world

Although aquaculture of kelp is relatively common around the world (Buschmann et al. 2017; Sondak et al. 2017), attempts to restore kelp forests and their associated communities are rare. Where it has been attempted, projects have focussed on restoration of kelp forest ecosystem services (e.g. biodiversity, kelp-associated fisheries) lost due to anthropogenic stressors (California, USA, North 1976; Wilson & McPeak 1983; Ancona Italy, Perkol-Finkel et al. 2012; Jeju, South Korea, Yoon et al. 2014), offsetting habitat loss due to coastal development (California, USA, Carter et al. 1985; Ambrose 1994), and scientific research (Tokyo, Japan, Terawaki et al. 2001; Atacama, Chile, Westermeier et al. 2016). Overall, success of restoration projects has been mixed (e.g. Carter et al. 1985; Ambrose 1994; Baja, Mexico, Hernández-Carmona et al. 2000), but some projects have recovered small stable populations of kelp (e.g. North 1976; Wilson & McPeak 1983; Terawaki et al. 2001; Reed et al. 2006, 2017).

Restoration of kelp habitats typically follows two broad strategies, assisted recovery and active restoration. Assisted recovery – where natural kelp recovery is facilitated by the removal of the agent of decline (e.g. culling of sea urchins, Ling 2008; Burdick et al. 2015) or the installation of substrata for kelp colonisation (e.g. artificial reefs, Carter et al. 1985; Ambrose 1994; Terawaki et al. 2001; Reed et al. 2006) – has been successful at increasing kelp recruitment over the short-term. However, results from these actions are highly variable and dependent on local conditions, and projects involving removal of the agents of decline have generally had greater success than those that provide artificial substratum for kelp recruitment alone. Critically, the efficacy of assisted recovery is often hindered by resource constraints (see Section 3.8), and by hysteresis effects that impair kelp recruitment/reestablishment even after the initial agent of decline has been ameliorated (Gorman & Connell 2009; Steneck & Johnson 2014). Indeed, we are aware of only one example where assisted recovery in isolation has resulted in long-term restoration of kelp (see Reed et al. 2006; 2017).

Active restoration efforts have had greater success, and typically involves transplanting of adult and/or juvenile kelp from a pre-existing donor site or outplanting of lab-cultured kelp (North 1976; Hernández-Carmona et al. 2000; Perkol-Finkel et al. 2012; Yoon et al. 2014; Westermeier et al. 2016). The long-term success of this approach is reliant on either ongoing transplantation of kelp, which can be cost-prohibitive and is dependent on a healthy donor population (North 1976; Devinny & Leventhal 1979), or adequate natural recruitment of juvenile kelp. In this latter instance, the recruitment source may be nearby populations of kelp or from the transplanted adult kelp. Overall, the emphasis is that recruitment of juvenile kelp and the continuation of self-sustaining generations is critical to long-term restoration success (see Operation Crayweed in Section 3.7). Of note, is that planting of juvenile kelp (whether lab-cultured or otherwise) has had little success (but see Perkol-Finkel et al. 2012) unless it is combined with concomitant planting of adult kelp (North 1976; Devinny & Leventhal 1979; Layton et al. in review). This may be due to increased herbivory, competition, or stressors that cause mortality of juvenile kelp in the absence of adults (Wood 1987; Hernández-Carmona et al. 2000; Konar & Estes 2003; Vergés et al. 2016; Layton et al. in review). Ultimately, the best results of kelp forest restoration seem to occur when a combination of techniques are used to achieve natural and ongoing recruitment of kelp.

## 3.4 Australian role of kelp forests

The role of kelp forests in Australia mirrors their role globally, where kelp form spatially complex habitats that support diverse and productive communities of flora and fauna (Bennett et al. 2016; Coleman & Wernberg 2017). Unlike the northern hemisphere where kelp canopies are primarily comprised of ‘true’ laminarian kelps, in Australia canopy-forming species are both laminarian and fucalean (Table 3.1), with a much larger diversity of subtidal fucoids than laminarian species (Womersley 1987; Phillips 2001; Coleman & Wernberg 2017). Kelp dominate most shallow (<30–50 m) rocky reefs in temperate Australia, and in a similar way to the Great Barrier Reef, these reefs form an integrated reef system called the Great Southern Reef (GSR, Bennett et al. 2016). This ~8,000 km long system of reefs is largely defined by the distribution of kelp forests in Australia, and spans from the Queensland/New South Wales border (~28.5° S), down the east coast of Australia and around Tasmania, along the continent’s southern coastline, and as far north as Geraldton, Western Australia (~29° S). Across this distance and in close proximity to the GSR, live more than 70% of Australia's population (i.e. ~17 million people) (Bennett et al. 2016).

As the foundation of the GSR, kelp forests in Australia support high levels of biodiversity and productivity. Kelp forests also have high economic value and support many commercial fisheries including the rock lobster and abalone fisheries, which are the nation's two most valuable fisheries and together contribute more than $500 million p.a. to the Australian economy (Bennett et al. 2016). Australian kelp forests are also home to species of high conservation value (e.g. eastern blue groper, and weedy and leafy seadragons), many of which are found nowhere else. Indeed, a remarkable feature of the biodiversity on the GSR is the high level of endemism. This is particularly true for seaweeds, and the GSR is considered a global hotspot of seaweed endemism (Philips 2001; Kerswell 2006). For example, more than 75% of the 600+ species of red seaweed found on the GSR are endemic to the region (Womersley 1994, 1996, 1998; Phillips 2001). More broadly, the GSR is also considered a biodiversity hotspot for sponges, crustaceans, chordates, bryozoans, echinoderms and molluscs, and rates of endemism within these taxa range from 20–60% (Bennett et al. 2016). Beyond the direct economic contribution of commercial fisheries, a lack of data makes it difficult to quantify the value of ecosystem services provided by Australian kelp forests. But nutrient cycling, carbon sequestration, and coastal protection (i.e. dampening of swells and lessening of erosion) are all critical yet undervalued ecosystem services provided by kelp forests (discussed further in Section 3.11) (Costanza et al. 1997; Smale et al. 2013; Bennett et al. 2016; Krause-Jensen et al. 2018). Indirect commercial and social benefits arising from kelp forests are also likely to be substantial, especially in coastal communities. These include indirect effects on fisheries (e.g. trophic linkages that influence coastal food webs and prey species), recreational fishing (worth ~A$500 million p.a.), ecotourism and other forms of marine recreation (e.g. whale watching, scuba diving) (Bennett et al. 2016).

## 3.5 Australian status of kelp forest

As outlined in Section 3.2, multiple regions in Australia have experienced significant declines in kelp over the past several decades (Wernberg *et al.* 2011; Bennett *et al.* 2016; Krumhansl *et al.* 2016; Evans *et al.* 2017), including areas of South Australia (Connell *et al.* 2008), Tasmania (Ling 2008; Johnson *et al.* 2011), Victoria (Jung *et al.* 2011), New South Wales (Andrew and O'Neill 2000; Vergés *et al.* 2016) and southern Queensland (Phillips and Blackshaw 2011). Additional work has also revealed significant losses of kelp in Western Australia, where a sustained marine heatwave over the summer of 2010–2011 in combination with southward range extension of subtropical herbivorous fishes associated with ocean warming, resulted in the permanent loss of forests of *Ecklonia radiata* (the dominant kelp on the GSR) from ~100 km of coastline (~2300 km$^2$) between Kalbarri and Geraldton (Wernberg *et al.* 2016).

In South Australia, kelp forest losses have been mostly attributed to urbanisation and increased runoff of sediments, nutrients and pollution (Connell *et al.* 2008; Gorman and Connell 2009). The result is that kelp forests within ~25 km of Adelaide, consisting mostly of *E. radiata*, have been largely replaced by less complex and less productive turf algae habitats (Figure 3.1). There has also been widespread loss of *E. radiata* kelp forests in Port Phillip Bay, Victoria, particularly along the western and northern coastlines near the metropolitan areas of Geelong and Melbourne (Jung *et al.* 2011; Kriegisch *et al.* 2016; S. Swearer *pers. obs.*). Here, overgrazing by sea urchins (*Heliocidaris erythrogramma*) is typically the proximal and initial cause of kelp destruction, but subsequent proliferation of turf algae, in part due to high nutrient loading, also continues to inhibit kelp recruitment (Kriegisch *et al.* 2016).

Destructive grazing by urchins is also a significant driver of kelp forest loss in Tasmania and the Bass Strait (Ling 2008; Johnson *et al.* 2011). Urchin barrens formed by the long-spined urchin (*Centrostephanus rodgersii*) are now extensive across these areas and have replaced the formerly lush kelp forests. The long-spined urchin was previously only characteristic of the New South Wales coast but has undergone southern range extension over the last several decades due to increasing poleward penetration of the East Australia Current (EAC, Ridgway 2007; Ling 2008; Johnson *et al.* 2011). While changing oceanography is responsible for the incursion of the urchin into Tasmanian waters, the proliferation of their populations is largely attributed to ecological overfishing of large southern rock lobster (*Jasus edwardsii*), which is the primary predator of *C. rodgersii* urchins in Tasmania (Ling *et al.* 2009; Johnson *et al.* 2011). Overgrazing by urchins has mostly affected *E. radiata* kelp forests, but Tasmania has also suffered extensive losses of giant kelp (*Macrocystis pyrifera*) forests (Johnson *et al.* 2011). While the loss of these iconic underwater forests (Figure 3.2) is mostly attributed to increasing influence of the warm, nutrient-poor waters of the EAC, it is likely that urchin overgrazing contributed to the problem in some areas (also see Sanderson 2003; Ling 2008). Moreover, expanding urchin barrens are certainly precluding any chance of giant kelp recovery in many areas of eastern Tasmania. Overall, more than 90% of Tasmania's giant kelp forests (which also occur to a lesser extent in parts of Victoria and South Australia) have been lost over recent decades, to be replaced by urchin barrens or *E. radiata* kelp forests (Johnson *et al.* 2011; Ling *et al.* 2015). As a consequence of these losses, in 2012 the giant kelp forests of southeast Australia became the first (and to date the only) marine community listed as threatened under the EPBC Act (see Section 3.6; Evans *et al.* 2017).

The long-spined urchin has also contributed to extensive losses of kelp forests across its native range in New South Wales (Andrew 1993; Andrew and O'Neill 2000). Indeed, *C. rodgersii* urchin barrens are estimated to extend across more than 50% of the shallow rocky reef habitats along the central and southern coastlines of the state (Andrew and O'Neill 2000), suggesting widespread losses of the two dominant kelp, *E. radiata* and *Phyllospora comosa*. These urchin barrens have formed over many decades, possibly due to overfishing of urchin predators – such as eastern rock lobster (*Sagmariasus verreauxi*) and eastern blue groper (*Achoerodus viridis*) (Ling *et al.* 2009, 2015; Ling and Johnson 2012; Evans *et al.* 2017) - and persistent declines in kelp cover in some regions due to urbanisation and increasing ocean temperatures (Andrew and O'Neill 2000; Coleman *et al.* 2008; Mabin *et al.* 2013).

Figure 3.1: *Ecklonia radiata* kelp forests (left) replaced by turf algae (right). Photos by Sean Connell, and reproduced under Creative Commons by Attribution from Connell *et al.* 2008, Marine Ecology Progress Series, Inter Research.

Figure 3.2: Photos of formerly healthy giant kelp (*Macrocystis pyrifera*) forests (left) and now degraded (right) forests taken at the same location in south-east Tasmania. Photos reproduced with permission of Matthew Ramaley (left) and Matthew Doggett (right).

Nearshore untreated sewage outfalls have been implicated in the local extinction of *P. comosa* kelp forests from the metropolitan coast around Sydney throughout the 1980s (Coleman *et al.* 2008). However, these forests are now being restored under the aegis of Operation Crayweed (see Section 3.7). Further losses of *E. radiata* kelp forests have occurred on the northern coasts of New South Wales, attributed to warming waters and overgrazing caused by increasing abundances of herbivorous subtropical fish species (Vergés *et al.* 2016). The poleward shift of subtropical species into temperate waters is referred to as tropicalisation and is expected to increase in the future as oceans continue to warm (Vergés *et al.* 2016; Pecl *et al.* 2017; Zarco-Perello *et al.* 2017).

Overall, Australian kelp communities are understudied relative to many tropical marine ecosystems, and analyses of kelp populations in many regions is hampered by a lack of data (Bennett *et al.* 2016; Krumhansl *et al.* 2016). Nonetheless, it seems clear that increasing ocean temperatures - especially in south-eastern and Western Australia, which are global hotspots of ocean warming (Hobday and Pecl 2014) - are likely to cause continued poleward range contractions of all kelp species in Australia, to be replaced by smaller subtropical seaweeds (e.g. Wernberg *et al.* 2016; Coleman *et al.* 2017).

## 3.6 Kelp forests and Matters of National Environmental Significance

The primary MNES related to the conservation and restoration of Australian kelp forests is listed threatened species and ecological communities. As a consequence of dramatic declines in the last few decades, the giant kelp (*Macrocystis pyrifera*) forests of southeast Australia became the first, and to date only, marine community listed as threatened under the EPBC Act. Listed in 2012, there is currently no recovery plan prepared for this threatened community (Evans *et al.* 2017). Numerous endemic fishes that live in and around Australian kelp forests are also listed as threatened species under the EPBC Act, including the spotted handfish (*Brachionichthys hirsutus*, critically endangered), red handfish (*Thymichthys politus,* critically endangered), Ziebell's handfish (*Brachiopsilus ziebelli*, vulnerable), black rockcod (*Epinephelus daemelii*, vulnerable) and the Syngnathidae family, which comprises over 110 species of Australian seadragons, seahorses and pipefish (many of which only occur in Australia's kelp forests). Numerous other species listed under the EPBC Act that live and forage in temperate nearshore waters of Australia are likely to rely at some level on kelp forest productivity and trophic linkages. These include white-bellied sea-eagles (*Haliaeetus leucogaster*, protected), grey nurse sharks (*Carcharias taurus*, critically endangered), white sharks (*Carcharodon carcharias*, vulnerable), and fur seals and sea lions (family Otariidae, protected). Along these same lines, the importance of kelp forests to migratory species that spend extended periods feeding in temperate nearshore waters of Australia should also be given consideration. Likewise, despite a lack of data and habitat mapping, it should be recognised that kelp forest habitats very likely occur on many shallow rocky reefs (<30–50 m) in temperate and subpolar latitudes within Commonwealth Marine areas - another MNES.

Table 3.1: The primary species/genera of habitat-forming kelp in Australia. References: 1. Edgar, G. J. 2008. Australian Marine Life: The Plants of Animals of Temperate Waters. 2nd edition. New Holland Publishers, Sydney; 2. Womersley, H. B. S. 1987. The Marine Benthic Flora of Southern Australia Part II. South Australian Government Printing Division, Adelaide; 3. Huisman, J. M. 2000. Marine Plants of Australia. University of Western Australia Press, Nedlands, WA; 4. Schiel, D. R., and M. S. Foster. 2015. The biology and ecology of giant kelp forests. University of California Press, Oakland, California; 5. South, P. M., O. Floerl, B. M. Forrest, and M. S. Thomsen. 2017. A review of three decades of research on the invasive *kelp Undaria pinnatifida* in Australasia: An assessment of its success, impacts, and status and one of the world's worst invaders. Marine Environmental Research 131:243-257.

| Species (Order) | Common name | Distribution | Description | Refer-ences |
|---|---|---|---|---|
| *Cystophora spp.* (Fucales) | cystophora | Sheltered to exposed reefs, 0–48 m. Nikol Bay, WA to Port Stephens, NSW and around TAS. | Grows to 4m. A widespread and highly diverse genus of kelp found only in Australasia. Can be locally abundant and dominant. May rise vertically (due to air-filled floats) or lay across the substrata. | 1, 2, 3 |
| *Durvillaea potatorum* (Fucales) | bull kelp | Exposed reef, 0–30 m. Cape Jaffa, SA to Tathra, NSW and around TAS. | Grows to 8m. A massive, thick and leathery kelp that lays prostrate across the substrata. The dominant species around low-tide level on exposed coastlines. | 1, 2, 3 |
| *Ecklonia radiata* (Laminariales) | common kelp, golden kelp | Moderately exposed reef, 0–60 m. Geraldton, WA to Brisbane, QLD and around TAS. | Grows to 1.5m. Most widespread and abundant kelp in Australia, with a distribution that mirrors the extent of the GSR. Very often the dominant kelp on the reef. Has a long rigid stipe (i.e. 'stem') that holds the fronds above the substrata. | 1, 2, 3 |
| *Lessonia corrugata* (Laminariales) | strapweed | Exposed reef, 0–20 m. Phillip Island, VIC and around TAS. | Grows to 1.5m. Occasionally locally abundant and dominant, typically in shallower and more exposed locations than *E. radiata.* Typically lies across the substrata. | 1, 2, 3 |
| *Macrocystis pyrifera* (Laminariales) | giant kelp, string kelp | Moderate to exposed reef, 0–28 m. Cape Jaffa, SA, to Walkerville, VIC and around TAS. | Grows taller than 40m. Has air-filled floats and can form immense underwater forests, often with a floating surface-canopy. Can be locally abundant and dominant. Has a shorter ecotype (~10 m, form *angustifolia*) that typically grows in shallower locations. | 1, 2, 3, 4 |
| *Phyllospora comosa* (Fucales) | crayweed | Moderate to exposed reef, 0–20 m. Robe, SA to Port Macquarie, NSW and around TAS. | Grows to 3m. Among the most common and dominant kelp on shallow and exposed sections of coastline. Has air-filled floats and typically lays just above the substrata. Often forms a dense band above the zone dominated by *E. radiata*. | 1, 2, 3 |
| *Sargassum spp.* (Fucales) | sargassum | Sheltered to exposed reefs, 0–48 m. Australia-wide | Grows to 1.5m. A diverse genus of kelp with global distribution that occur throughout tropical and temperate Australia. Can be locally abundant and dominant. May rise vertically (due to air-filled floats) or lay across the substrata. | 1, 2, 3 |
| *Scytothalia dorycarpa* (Fucales) | western crayweed | Moderate to exposed reef, 0–44 m. Geraldton, WA to Point Lonsdale, VIC. | Grows to 2m. Fulfils a similar role to *P. comosa,* especially in Western Australia. | 1, 2, 3 |
| *Undaria pinnatifida* (Laminariales) | Japanese kelp | Moderately exposed reef, 0–10 m. Port Phillip and Apollo Bays, VIC, and D'Entrecasteaux Channel to Coles Bay, TAS. | Grows to 1m. An introduced and invasive species. Occasionally locally common and dominant but highly seasonal, almost disappearing throughout summer and autumn. Has a rigid stipe (i.e. 'stem') that holds the fronds above the substrata. | 1, 3, 5 |

## 3.7 Success and failure of kelp forest restoration in Australia

There have been few attempts to restore kelp forests in Australia. The earliest documented work comes from the Seacare community group (Sanderson 2003) and describes attempts to restore giant kelp (*Macrocystis pyrifera*) in Tasmania. Several techniques were used, including transplanting juvenile kelp from donor populations; transplanting substrata on which juvenile giant kelp were growing following natural recruitment; transplanting sporophylls (i.e. the reproductive fronds of giant kelp); and outplanting lab-cultivated juvenile giant kelp (~5 mm in length). Long-spined urchins (*Centrostephanus rodgersii*) were also removed at some restoration sites and improved the chance of positive outcomes. However, the project realised only limited success, and outcomes varied markedly across sites. A single small patch of giant kelp was established at one site but subsequently disappeared over several seasons as part of the ongoing decline of giant kelp in southeast Australia. The methods employed at this site did not differ to those at other sites (i.e. transplanting ~100 juvenile kelp and three fertile sporophylls), but this site was the most exposed and southerly of the locations (i.e. furthest from the east and north-east regions of Tasmania that experienced the greatest declines in giant kelp cover) and had the most similar community composition to the donor site.

Operation Crayweed is the only other reported example of targeted kelp forest restoration in Australia of which we are aware. This ongoing project begun in 2012 with aims to restore crayweed (*Phyllospora comosa*) forests to metropolitan Sydney where they were once abundant (Campbell et al. 2014; Marzinelli et al. 2014, 2016, in prep.; OperationCrayweed.com). Adult crayweed are being transplanted from donor populations outside of metropolitan Sydney to restoration sites, with the primary aim of establishing sufficient adult individuals to provide recruitment of juvenile crayweed. Despite high site variability, survival of transplanted adult crayweed was typically comparable to natural mortality (Campbell *et al.* 2014), and at two of the initial sites, transplanted crayweed have reproduced so that multiple generations are now identifiable. At these sites, mature individuals can now be found hundreds of metres from the original patches established in 2012 (Marzinelli *et al.* unpublished data). These crayweed forests have rapidly become self-sustaining with no additional cost or maintenance, which is a rare result in marine restoration. Critically, this relatively small-scale intervention has translated into a large-scale impact, with crayweed populations continuing to expand and colonise substantial areas and beginning to function as natural forests (Marzinelli *et al.* 2016, Marzinelli *et al.* unpublished data)

Additional research has employed aspects of active restoration and assisted recovery to improve understanding of kelp forests and ecological restoration. Gorman & Connell (2009) illustrated that kelp recovery can occur following removal of the turf algae that flourished and replaced kelp forests on reefs around Adelaide due to increased nutrient run-off. Several studies have also illustrated that removal of urchins can facilitate recovery of kelp and other seaweeds on Australian temperate reefs (Fletcher 1987; Ling 2008; Ling *et al.* 2010). Work by Layton *et al.* (in review) demonstrated the successful transplanting of over 500 adult *Ecklonia radiata* onto artificial reefs spread over 1.5 ha in Tasmania. Survivorship of transplants was comparable to natural reefs, and abundant recruitment of juveniles (>750) ensured that many patches become self-sustaining. Additionally, this work demonstrates the importance of maintaining/creating patches of kelp above a critical size threshold to ensure adequate recruitment of juvenile kelp, and thus habitat stability.

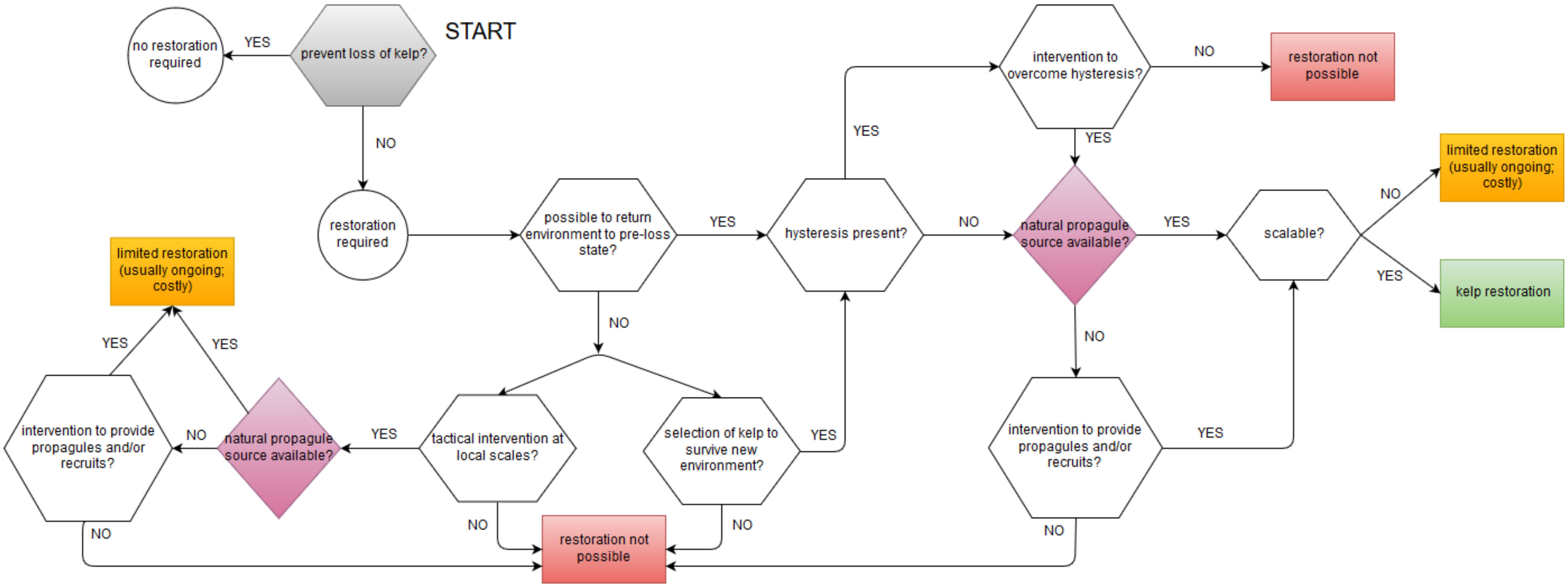


Figure 3.3. A workflow to determine if kelp restoration is possible and if so what approaches are most appropriate.

## 3.8 Recent advances and new ideas for kelp forest restoration

The loss of kelp forests in Australia is complex due to the multitude of different stressors and high levels of geographic variation (see Section 3.5). Accordingly, we developed a workflow as a useful approach to restoration of these complex systems, while also providing a summary of potential local outcomes (Figure 3.3). Most critical, the workflow illustrates that there are multiple pathways of restoration along with multiple endpoints, including circumstances where restoration is not possible or advisable (also see Johnson *et al.* 2017). This multiplicity, both of pathways and endpoints, exists because of environmental factors that are beyond the control of restoration practitioners, such as whether hysteresis is present in the pre-restoration system (e.g. an urchin barren, Johnson *et al.* 2017). The diverse pathways of kelp forest restoration is also reflective of variations in the driver of decline, the resources available for restoration efforts, and the scalability of the intervention. The design of the workflow is also indicative that at each decision node, science is needed to inform evidence-based decision-making and progression to the next stage. Adopting this workflow should help ensure restoration efforts are effective within resource constraints and that, critically, the agent of kelp forest decline has been addressed.

Briefly, progressing through the workflow towards the point of successful kelp restoration illustrates several key decision nodes. Firstly, is it possible to return the environment to its pre-loss state? If not, intervention is required to select and facilitate kelp to survive in the new environmental state. An example of this might be the selective breeding of thermally tolerant kelp from remaining healthy individuals and acclimation of early life history stages to increase thermal tolerance. If it is not possible to ameliorate or adapt to the novel ecosystem state, it seems that restoration efforts will, at best, be limited and at the tactical scale. Secondly, is there hysteresis present in the system after it has been returned to the pre-loss state? Such hysteresis effects can prove one of the biggest challenges to kelp forest restoration (Gorman and Connell 2009; Johnson *et al.* 2017). Lastly, for successful restoration to occur, efforts to overcome hysteresis and/or provide a novel source of propagules must be scalable, and commensurate with the scale of the initial degradation (Johnson *et al.* 2017; Marzinelli *et al.* in prep.).

## 3.9 Estimation of the costs of implementation

Estimating the costs of implementing effective kelp forest restoration is difficult considering there are so few Australian examples to date. For Operation Crayweed, workers transplanted approximately six 2 m$^2$ patches of crayweed (*Phyllospora comosa*) at each restoration site, with adult kelp densities of 15 m$^{-2}$.

Initial transplanting efforts at each site required ~five days and included site marking and preparation, securing of the mesh mats for crayweed attachment, collection of adult crayweed from the donor population, and the transplanting itself. Costs of these efforts are estimated at ~AU$10,000, and cover a four-person team, boat and tow-vehicle, SCUBA tank fills, basic equipment and consumables (Marzinelli *et al. unpublished data*). Project management and ongoing monitoring of the multiple Operation Crayweed sites is estimated at an additional AU$27,000 p.a. These costs do not include the science necessary to underpin decisions such as choice of donor site, size of patch, etc.

Assisted recovery techniques such as urchin culling have also produced promising results at local scale and will likely be a key consideration for many kelp restoration projects. However, urchin culling is a high-cost activity and is thus only suited to tactical control of urchins at small spatial scales and in order to maintain or bolster resilience of existing kelp habitats (Ling and Johnson 2012; Layton *et al.* in review), remove incipient barrens (Ling 2008; Tracey *et al.* 2015), or support active restoration efforts (Sanderson 2003). Economic projections indicate that culling of *Centrostephanus rodgersii* urchins from densities of 0.5 urchins $m^{-2}$ down to 0.1 urchins $m^{-2}$ (the approximate density required for kelp recovery) across a 1 $km^2$ area of reef and from depths of 0-20 m would take two divers 725 days and cost ~AU$750,000 (Tracey *et al.* 2014). While costs can be reduced considerably by culling across a narrower depth range, these projections are nonetheless conservative given that urchin densities on barrens commonly exceed 1 urchin $m^{-2}$ (Ling 2008; Ling and Johnson 2012). Novel technology is promising to improve the scalability and cost-effectiveness of urchin culling however, and trials of 'smart' autonomous underwater vehicles designed to locate and kill urchins are currently under way (Johnson *et al. unpublished data*). Overall, the impetus to consider restoration of kelp forests may benefit greatly from environmental accounting to ascertain the value of kelp forests to human society and underpin rigorous cost-benefit analysis (e.g. Rogers *et al.* 2018). This is especially true since it is likely possible to decrease the costs of underwater restoration operations substantially by reducing diving labour, and increasing automation and efficacy of mass seeding techniques (e.g. mass dispersal of lab-cultured kelp propagules from boat-mounted pumps, North 1976).

## 3.10 Matters of National Environmental Significance that could benefit from on-ground restoration investment

As the foundation of Australia's rocky reef ecosystems, increasing kelp forest health and abundance via restoration efforts is likely to result in concordant benefits to the associated community. Therefore, each MNES identified in Section 3.6 would benefit from investment in kelp forest restoration. Work has demonstrated that recovered *Ecklonia radiata* kelp forests following urchin removal support similar communities to natural *E. radiata* forests (Ling 2008). Conversely, while restoration of *Phyllospora comosa* kelp forests by Operation Crayweed has dramatically improved species richness at restoration sites, the restored community may still take some time to approach the same composition of natural *P. comosa* forests (Marzinelli *et al.* 2016). Thus, although restoration of key habitat-forming species can aid recovery of the associated community, restoration of the original biodiversity associated with these habitats can be a complex and long-term process (see also Reed et al. 2017).

## 3.11 Other benefits from kelp forest restoration

Coastal seaweeds beds – of which kelp are the largest component by biomass – have been identified as important marine sinks of carbon (so called Blue Carbon; Duarte 2017; Krause-Jensen *et al.* 2018). Moreover, a substantial portion of the carbon assimilated by kelp forests is sequestered away in coastal sediments and the deep ocean (Krause-Jensen and Duarte

2016; Filbee-Dexter and Wernberg 2018). Indeed, estimates of carbon sequestration by kelp-dominated seaweed beds rival or even exceed that achieved by other plant-based coastal habitats such as seagrass meadows or mangroves (Krause-Jensen and Duarte 2016; Krause-Jensen *et al.* 2018).

Similarly, kelp have a great ability to absorb nitrogen and other coastal nutrients to allow for bioremediation. Integrated Multi-Trophic Aquaculture (IMTA) is a rapidly emerging field and can utilise kelp to absorb and offset excess nutrients associated with shellfish or finfish aquaculture (Buschmann *et al.* 2017; Hadley *et al.* 2018). Emerging technologies and investment are also positioning kelp and other seaweeds as a cornerstone of blue economy applications, including as food for human consumption, livestock feed, biofuel, nutraceuticals, and pharmacological applications (Buschmann *et al.* 2017; Sondak *et al.* 2017).

Kelp forests also modify local hydrography, and can bolster coastal defences by dampening ocean swell and decreasing erosion (Jackson and Winant 1983; Gaylord *et al.* 2007; Løvås and Tørum 2001; Smale *et al.* 2013). Thiese benefits should be given special consideration with regards to forecast increases in sea level and storm activity due to anthropogenic climate change (Intergovernmental Panel on Climate Change 2014).

Kelp and associated seaweeds also play an important role in Indigenous Australian culture and tradition (Thurstan *et al.* 2018). Contemporary and historical uses include ceremonial activities, medicinal uses, clothing, food, shelter and as domestic devices. Archival records of the use of bull kelp (*Durvillaea potatorum*) by Indigenous Australians are particularly numerous, and there is considerable contemporary use of this kelp by Indigenous practitioners in artistic and knowledge-sharing activities (Thurstan *et al.* 2018). Recognising this traditional knowledge creates opportunities to conserve and revitalise traditional customary practices and Indigenous business activities. Furthermore, the culturing, outplanting and monitoring that large-scale kelp forest restoration efforts require provide ideal opportunities for Indigenous employment, management, ownership and, to establish the skills and knowledge that underpin seaweed farming.

## 3.12 Potential indicators to be used in cost-effectiveness and subsequent monitoring of outcomes

Since kelp are the foundation of threatened kelp forest communities and inhabitants, comparison of community composition between restored and natural ‘reference sites’ should provide robust indicators of restoration success relative to MNES. The Society of Ecological Restoration has also developed a rigorous system of International Standards for the practice of ecological restoration (see McDonald et al. 2017). Central to the standards is the so-called “5-star recovery system”, which uses six key ecosystem attributes to assess restoration projects and measure progress along a trajectory of recovery (Table 3.2). This provides a well-established framework upon which kelp forest-specific indicators and metrics can be developed, which may include transplant survival, growth rates, condition (e.g. fouling, bleaching, photosynthetic efficiency), genetic diversity, and recruitment. Certainly, recruitment of juvenile kelp is one of the greatest indicators of ongoing success and kelp forest resilience. Ultimately, the ideal goal as demonstrated by Operation Crayweed, is kelp forest recovery and reestablishment beyond the restoration footprint due to overflow of natural recruitment.

Table 3.2: Society of Ecological Restoration's 1-5 star recovery scale interpreted in the context of the six key ecosystem attributes used to measure progress towards a self-organizing status. Reproduced from McDonald *et al.* 2016 with permission of the Society of Ecological Restoration

| ATTRIBUTE | ★ | ★★ | ★★★ | ★★★★ | ★★★★★ |
|---|---|---|---|---|---|
| **Absence of threats** | Further deterioration discontinued and site has tenure and management secured. | Threats from adjacent areas beginning to be managed or mitigated. | All adjacent threats managed or mitigated to a low extent. | All adjacent threats managed or mitigated to an intermediate extent. | All threats managed or mitigated to high extent. |
| **Physical conditions** | Gross physical and chemical problems remediated (e.g., contamination, erosion, compaction). | Substrate chemical and physical properties (e.g., pH, salinity) on track to stabilize within natural range. | Substrate stabilized within natural range and supporting growth of characteristic biota. | Substrate securely maintaining conditions suitable for ongoing growth and recruitment of characteristic biota. | Substrate exhibiting physical and chemical characteristics highly similar to that of the reference ecosystem with evidence they can indefinitely sustain species and processes. |
| **Species composition** | Colonising native species (e.g., ~2% of the species of reference ecosystem). No threat to regeneration niches or future successions. | Genetic diversity of stock arranged and a small subset of characteristic native species establishing (e.g., ~10% of reference). Low onsite threat from exotic invasive or undesirable species. | A subset of key native species (e.g., ~25% of reference) establishing over substantial proportions of the site. Very low onsite threat from undesirable species. | Substantial diversity of characteristic biota (e.g. ~60% of reference) present on the site and representing a wide diversity of species groups. No onsite threat from undesirable species. | High diversity of characteristic species (e.g., >80% of reference) across the site, with high similarity to the reference ecosystem; improved potential for colonization of more species over time. |
| **Structural diversity** | One or fewer strata present and no spatial patterning or trophic complexity relative to reference ecosystem. | More strata present but low spatial patterning and trophic complexity, relative to reference ecosystem. | Most strata present and some spatial patterning and trophic complexity relative to reference site. | All strata present. Spatial patterning evident and substantial trophic complexity developing, relative to the reference ecosystem. | All strata present and spatial patterning and trophic complexity high. Further complexity and spatial pattering able to self-organize to highly resemble reference ecosystem. |
| **Ecosystem functionality** | Substrates and hydrology are at a foundational stage only, capable of future development of functions similar to the reference. | Substrates and hydrology show increased potential for a wider range of functions including nutrient cycling, and provision of habitats/resources for other species. | Evidence of functions commencing - e.g., nutrient cycling, water filtration and provision of habitat resources for a range of species. | Substantial evidence of key functions and processes commencing including reproduction, dispersal and recruitment of species. | Considerable evidence of functions and processes on a secure trajectory towards reference and evidence of ecosystem resilience likely after reinstatement of appropriate disturbance regimes. |
| **External exchanges** | Potential for exchanges (e.g. of species, genes, water, fire) with surrounding landscape or aquatic environment identified. | Connectivity for enhanced positive (and minimized negative) exchanges arranged through cooperation with stakeholders and configuration of site. | Connectivity increasing and exchanges between site and external environment starting to be evident (e.g., more species, flows etc.). | High level of connectivity with other natural areas established, observing control of pest species and undesirable disturbances. | Evidence that potential for external exchanges is highly similar to reference and long term integrated management arrangements with broader landscape in place and operative. |

# 4 SHELLFISH REEFS

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## 4.1 Global role of shellfish reefs

Some bivalve species form complex, three-dimensional habitats made up of dense aggregations of bivalves, their shells, associated species, and accumulated sediments. These ecosystems were generally engineered by oyster (generally referred to as reefs) or mussel (generally referred to as beds) species. In this report, we will refer to these ecosystems as shellfish reefs. Shellfish reefs were a dominant habitat type in temperate and subtropical estuaries and other sheltered coastal environments around the world (Beck 2009). For example, oyster reefs were so extensive in estuaries on the Atlantic and Gulf Coasts of the USA that they were considered to be a navigation hazard (Coen and Grizzle 2007) and may have dominated over a quarter of the Willapa Bay (Washington State, USA) bottom on the Pacific coast Blake and Zu Ermgassen 2015). Historical accounts describe natural oyster beds stretched over a distance of thirty miles in length and from four to seven in width (Bancroft 1890). The most extensive oyster grounds surveyed in North America included 25,500 ha in Tangier and Pocomoke Sounds (Chesapeake Bay, Virginia, USA) in 1878 and 16,500 ha in Matagorda Bay, Texas (Zu Ermgassen *et al.* 2012). In Europe, over a century ago, one-fifth of the Dutch part of the North Sea was covered with European oyster (*Ostrea edulis*) beds (Gercken and Schmidt 2014).

These ecosystems supported major harvest fisheries targeting their food value and their shells, which were used for lime production and other construction materials. The mid to late 1800s marked peak harvest years in Europe and North America and the rapid devastation of many reefs and beds. The scale of these harvests were vast. For example, 700 million oysters were consumed in London in 1984 and up to 120,000 men were employed in the British oyster dredge fishery (Kennedy *et al.* 2011). In the Chesapeake Bay, USA peak harvest during the 1880s reached 20 million bushels of oysters annually (2 billion oysters or 900,000 tonnes).

Recently, benefits of shellfish reefs other than as a fishery resource have been recognised, including their role in boosting local fish and crustacean fisheries, improving local water quality, and protecting shorelines (zu Ermgassen *et al.* 2016; McLeod *et al.* 2019). The economic value of the full suite of ecosystem services derived from natural oyster reefs in North America was recently estimated to be as high as US$99,000 ha$^{-1}$ year$^{-1}$ (Grabowski *et al.* 2012), which is higher than estimates for other habitats such as mangroves (Balmford *et al.* 2002), seagrass (Grabowski *et al.* 2012) and permanent wetlands (Sutton and Costanza 2002).

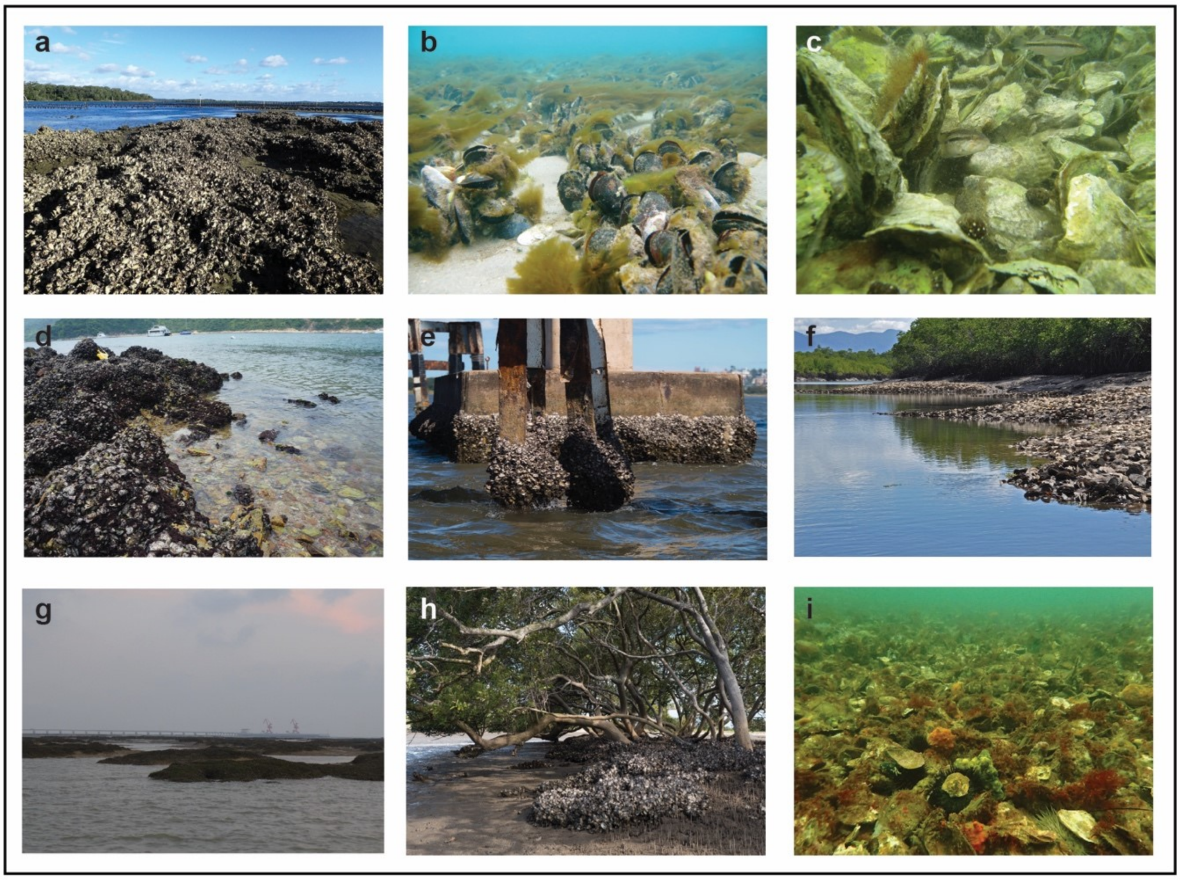


Figure 4.1: Shellfish habitats. (a) Intertidal Sydney rock oyster, *Saccostrea glomerata*, growing on a mud bank in Port Stephens, New South Wales, Australia. Photo by I. McLeod. (b) Subtidal green-lipped mussel, *Perna canaliculus*, bed growing on sand in an estuary channel in the Hauraki Gulf, New Zealand. Photo by I. McLeod. (c) Subtidal eastern oyster, *Crassostrea virginica*, with juvenile black sea bass, *Centropristis striata*, (located at the center of the image) Block Island, Rhode Island, USA. Photo by S. Brown. (d) Hooded oyster, *S. cucullata*, growing on a rocky shoreline, Hong Kong, China. Photo by D. McAfee. (e) *S. glomerata* growing on wharf pilings in Port Stephens, Australia. Photo by I. McLeod. (f) Leaf oyster, *Isognomon ephippium* growing on a mud bank in Hinchinbrook Channel, Queensland, Australia. Photo I. McLeod. (g) Liyashan Reef, made up of *Crassostrea sikamea* growing on mud flats, Jiangsu Province, China. Photo by J. Cheng. (h) *S. glomerata*, growing on mangrove roots and pneumatophore in Port Stephens, New South Wales, Australia. Photo by S. McOrrie. (i) Subtidal flat oyster, *A. angasi*, reef in Tasmania, Australia.

## 4.2 Global status of shellfish reefs

Shellfish reefs are threatened globally. For example, Beck *et al.* (2009; 2011) estimated that 85% of oyster reefs were lost globally and were functionally extinct (>99% loss) in 37% of estuaries. There has been little research into the status of other reef-forming shellfish species; however, when information is available this also points to widespread loss. For example, green-lipped mussel (*Perna canaliculus*) beds in New Zealand appear to occur at less than 1% of historical levels (McLeod 2009; Paul 2012). These losses are greater than those reported for other important estuary habitats including coral reefs, mangroves and seagrasses (Grabowski *et al.* 2012). The loss of this fishery resource has had devastating effects on the coastal communities that relied on the harvest of bivalve ecosystems for employment and food (McLeod *et al.* 2019). Through the process of historical amnesia, or shifting baselines, successive generations of local people and managers have grown accustomed to the new norm and have often forgotten about the former abundance of bivalve ecosystems (Beck 2009).

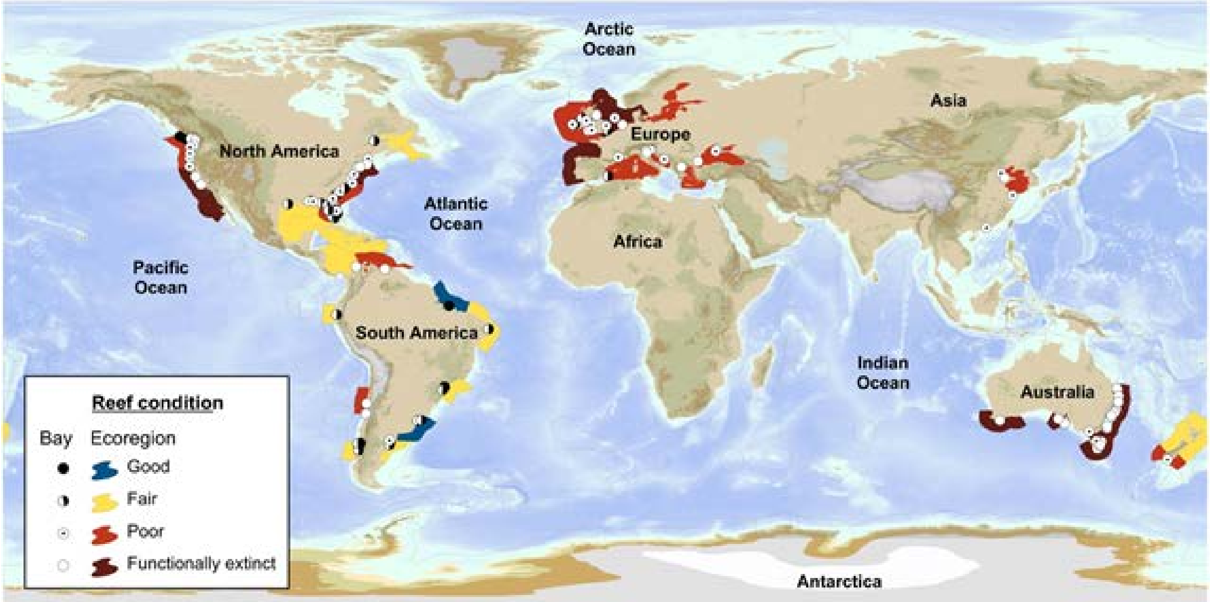


Figure 4.2: The global condition of oyster populations, with condition ratings based on the current abundance divided by the historical abundance of oyster reefs: < 50% lost (good); 50-89 % lost (fair); 90-99% lost (poor); > 99% lost (functionally extinct;). Adapted from Beck *et al.* (2011).

## 4.3 Success and failure of shellfish reef restoration around the world

Until recently, shellfish reefs were primarily managed as an important fisheries resource and restoration was focused on fisheries enhancement to support continued harvest (McLeod *et al.* 2019). It is likely that people have translocated shellfish to establish easily accessed populations for millennia and there is a blurred line between shellfish restoration, local enhancement and aquaculture. Large-scale reintroductions have taken place, sometimes between countries in areas where shellfish reefs fisheries collapsed. For example, between 1894 and 1930, large amounts of juvenile oysters from the Netherlands, France and Norway were distributed in the North Wadden Sea to restore beds for commercial fishing (Gercken and Schmidt 2014) and in the early 1880s, vast numbers of rock oysters, *Saccostrea glomerata* were shipped from New Zealand to Australia to restore local stocks (Ogburn *et al.* 2007). Although these translocations were sometimes successful in supporting fisheries over the short-term, they often created new problems with exotic diseases, competitors and predators introduced along with the oysters (Wolff and Reise 2002). Another strategy was the broad-scale placement of shell or shell fragments at high densities on the seafloor to create a new settlement surface. This led to a large-scale, and reasonably successful ‘put and take’ fishery in the USA, where shell was laid down on the seafloor to catch spat, then the oysters are dredged up once grown, and the cycle was repeated (Schulte 2017).

Since the 1990s, restoration efforts have been actioned for ecosystem benefits such as water quality improvements, shoreline protection and erosion control and providing food and habitat for harvested species (Brumbaugh and Coen 2009). Hundreds of bivalve restoration attempts have been made in the last three decades (Kennedy *et al.* 2011). Restoration attempts have generally tried to overcome one or both of the two main limiting factors inhibiting natural recovery, substrate and recruitment limitation (Figure 4.2).

In substrate-limited areas construction of settlement habitat using natural materials often involves placing bivalve shells sourced from the hospitality industry, dredged fossils or mined shells, directly on the seafloor (Brumbaugh and Coen 2009). This creates a recruitment surface for shellfish larvae, and provides structural complexity and habitat for a vast array of other species, even prior to live shellfish developing on the shells (Lehnert and Allen 2002). In projects where the primary objective is to restore ecosystem services provided by intact shellfish reefs (as opposed to restoring the reefs themselves), the shells are often placed in mesh bags, to minimise shell loss (Brumbaugh and Coen 2009). The use of bivalve shells for restoration is limited by the supply of shells (Luckenbach *et al.* 1999). Given the increased scarcity of shell materials, construction of settlement habitat using artificial materials has been increasing. These can range from products that are natural but not found in the shellfish reef environment, like concrete, crushed limestone and marl, to more artificial structures like recycled crab pots and clam cages, cement reef balls, crushed concrete, coal fly ash and broken porcelain fixtures (Chatry *et al.* 1986; Brumbaugh and Coen 2009; Kroeger 2012).

Attempts to restore populations with limited natural recruitment often use hatchery-produced bivalve spat settled onto shell or other materials. These are then deposited onto the seafloor or a constructed reef in an effort to overcome recruitment and post-settlement survival bottlenecks (Brumbaugh and Coen 2009). An alternative strategy is to reintroduce adult shellfish with the hope of establishing a breeding population. More recently, there has been a focus on breeding disease-resistant shellfish for restoration efforts (Lipcius *et al.* 2015).

Historically, the success of restoration projects has been mixed with many projects suffering from a lack of monitoring and poorly defined objectives (Kennedy *et al.*, 2011). Many restoration projects were also not protected from dredging leading to their failure (Schulte 2017). For restoration success, it is critical to establish what is limiting natural recovery prior to restoration so that the restoration approach can be tailored to that limitation. If there is a mismatch between the limitation and restoration method, it may results in reduced success of the projects. For example, a majority of restoration projects in Chesapeake Bay addressed substrate limitation by creating recruitment substrates, but failed to consider the availability of recruits to the sites (Brumbaugh and Coen 2009). In these systems, it may be more appropriate to use a combination of techniques that addresses the limits to natural recovery in the system. Projects that closely mimic the conditions of natural shellfish reef have proven to be the most successful in the long term. Often, this means elevating reefs above soft sediments to prevent smothering and water quality issues, establishing a large enough population or metapopulation and banning dredging and other destructive methods of harvesting restored sites (Lipcius *et al.* 2015).

Building on the success and failure of three decades of restoration trials, successful shellfish reef restoration has been scaling up, particularly in the USA. This has been led ‘top-down’ by large government initiatives and ‘bottom up’ by community groups. In 2004, the U.S. Army Corps of Engineers constructed a 42 ha oyster reef by placing dredged and washed oyster shells in Great Wicomico River, Chesapeake Bay. Schulte *et al.* (2017) reported the success of the restoration with 180 million oysters present, making this the largest wild oyster population in the world. The success of this restoration was attributed to the absence of dredge fishing and to the high vertical relief of the reefs, which mimicked historical natural reefs. The largest current initiative is the Chesapeake Bay Executive Order, which requires the oyster populations of 20 Chesapeake Bay tributaries to be restored by 2025. One of the target tributaries is Harris Creek, where between 2012 and 2016, 142 ha of oyster reefs were successfully restored, at a cost of US$28 million.

Efforts to develop local and regional bivalve habitat restoration plans have been increasing in North America and Europe in recent times and guidelines have been produced such as ‘A practitioners guide to the design and monitoring of shellfish restoration projects’ (Brumbaugh *et al.* 2006) and ‘Oyster habitat restoration. Monitoring and assessment handbook’ (Baggett *et al.* 2014). Tools such as the oyster calculator produced by The Nature Conservancy (oceanwealth.org/tools/oyster-calculator) that allows managers to calculate how much oyster restoration is needed to reach water quality and fish productivity goals are likely to further build the case for sustained investments in shellfish reef restoration.

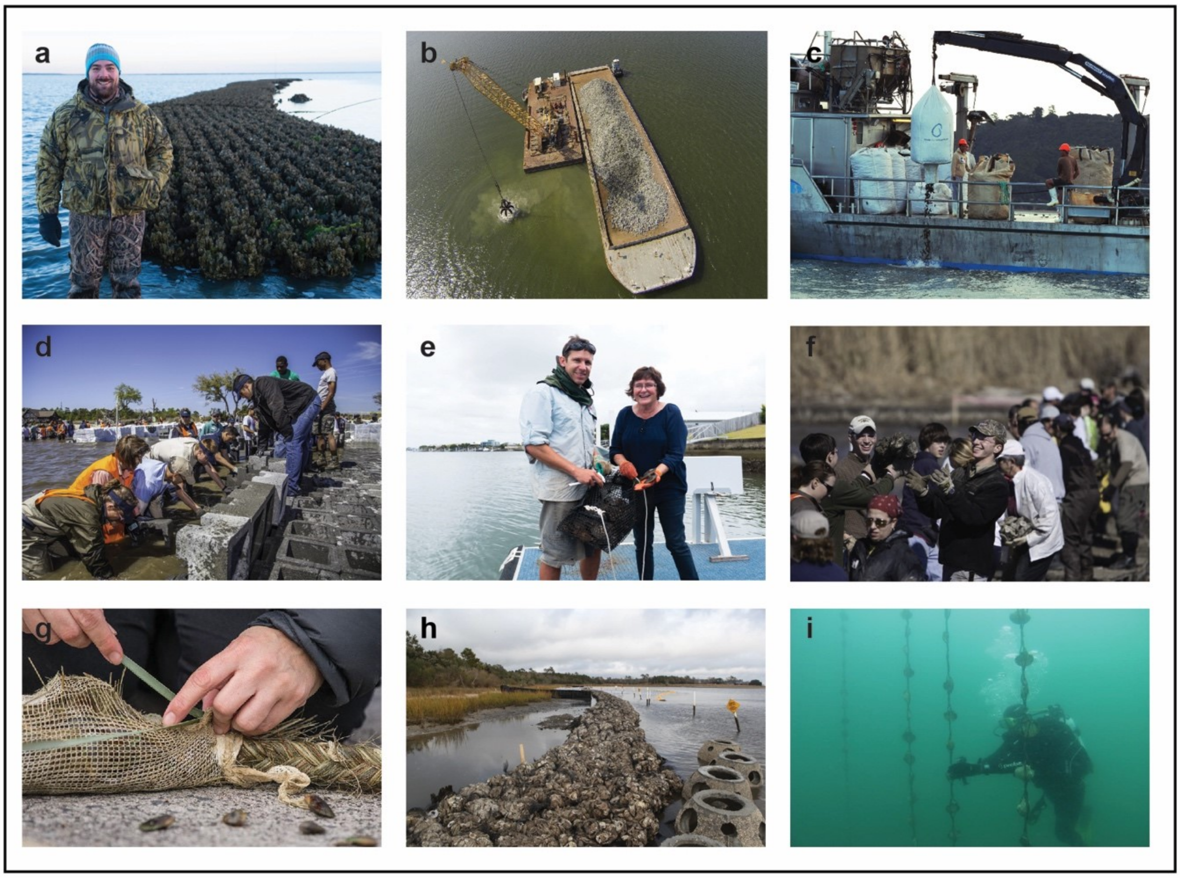


Figure 4.3: Shellfish ecosystem restoration. (a) Constructed oyster bank using oyster castles. Virginia, USA. Photo I. McLeod. (b) Granite rock being deployed as oyster settlement substrate in the Piankatank River, Virginia, USA. Photo U.S. Army /Patrick Bloodgood. (c) Live adult green-lipped mussels, *Perna canaliculus*, being deployed in to form beds in the Hauraki Gulf, New Zealand. Photo by Shaun Lee. (d) Volunteers assisting with oyster castle deployment, USA. Photo by Erika Norteman / The Nature Conservancy (e) Oyster gardeners with oyster basket, Queensland, Australia. Photo by Ian McLeod. (f) Volunteers moving bags of oyster shells for intertidal oyster restoration, USA. Photo by Erika Norteman / The Nature Conservancy. (g) Traditional Maori flax weaving being used to create mussel settlement substrate, Auckland, New Zealand. Photo by Shawn Lee. (h) Living shoreline in North Carolina with bagged oyster shell and reef balls deployed to provide a settlement substrate for oysters and to protect the shoreline from erosion. Photo by Jackeline M. Perez Rivera, U.S. Marine Corps photo. (i) *Ostrea angasi* spat being grown out on scallop shells prior to deployments in Port Phillips Bay, Australia. Photo by Ben Cleveland.

## 4.4 Australian role of shellfish reefs

### 4.4.1 Habitat provisioning

Shellfish reefs often provide the only hard substrate in otherwise soft sediment habitats As such, they provide structural complexity and hard substrates to which other sessile organisms can attach; they also provide refuge for prey, modify predator-prey interactions, act as a nursery for many marine organisms and trap sediments (Coen and Grizzle 2007; McLeod *et al.* 2014; zu Ermgassen *et al.* 2016). A wide range of organisms such as gastropods, crustaceans and polychaetes are found in native rock oyster reefs (Wilkie *et al.* 2012; McAfee *et al.* 2016; McLeod *et al. unpublished data*). The importance of oyster reefs as habitat providers was illustrated in a recent study where intertidal native rock oyster reefs were found to have a distinct assemblage of macroinvertebrates, with 30% higher densities (6,151 invertebrates $m^{-2}$), five times the biomass and almost five times the productivity of adjacent bare sediments (McLeod *et al.*, *unpublished data*). Similarly, native flat oyster reefs in Georges Bay, Tasmania, have been found to have three times the diversity and abundance of fauna than nearby soft sediments (Heller-Wagner 2017). Fish densities in intertidal native rock oyster reefs at high tide have been recorded at 1.0 fish $m^{-2}$ with the most abundant species belonging to the gobiid and blenniid families (McLeod *et al. unpublished data*).

### 4.4.2 Fisheries

The commercial value of fisheries supported by oyster reefs in the USA has been estimated to be US$4123 $ha^{-1}$ $yr^{-1}$ in 2011 (Grabowski *et al.* 2012). Research into the habitat value of Australian shellfish reefs and their role in supporting commercially and recreationally targeted fish and decapod species is just beginning in Australia. However, shellfish reefs were likely important foraging areas for snapper (*Pagrus auratus*; (Hamer *et al.* 2013), bream (*Acanthopargus* sp.), King George whiting (*Sillaginodes punctatus*), estuary perch (*Macquaria colonorum*), tailor (*Pomatomus saltatrix*) and tarwhine (*Rhabdosargus sarba*; (Gillies *et al.* 2015). Recent research using unbaited underwater cameras in New South Wales identified 35 fish species associated with remnant intertidal native rock oyster reefs when inundated. These included recreationally and commercially important fishes such as dusky flathead (*Platycephalus fuscus*), grey mullet (*Mugil cephalus*), yellowfin bream (*Acanthopagrus australis*), sand whiting (*Sillago ciliata*), luderick (*Girella tricuspidata*), common silverbiddy (*Gerres subfasciatus*), largetooth flounder (*Pseudorhombus arsius*), sand mullet (*Myxus elongatus*), and goldspot mullet (*Liza argentea*). The targeted decapod species mud crabs (*Scylla serrata*) and blue swimmer crabs (*Portunus pelagicus)* were also recorded in this study (Baena, *unpublished data*). Shellfish reefs in Australia generally support abundant and diverse macroinvertebrate communities (McAfee et al. 2016; Heller-Wagner 2017; McLeod *et al.*, *unpublished data*), which are likely to be an important food source for fish species. However, further research is required to determine the role that Australian shellfish reefs play in supporting food chains, fisheries production and fisheries aggregations.

### 4.4.3 Filter feeding and nutrient cycling

Bivalves are efficient filter feeders, which pull particulates from the surrounding water column, assimilate nutrients into their flesh and shell, and produce rich biodeposits (Newell and Mann 2012). In degraded ecosystems overseas where water quality is low and nutrient removal is a priority, the value of filtration by oyster communities (based on USA systems) was estimated at USD $1,385 – $6,716 $ha^{-1}$ $year^{-1}$ in 2011 (Grabowski *et al.* 2012).This filtration acts as an important bentho-pelagic coupler cycling nutrients between the plankton and benthic communities. This can be an important driver of benthic food chains. For example, the biodeposits of blue mussels (*Mytilus edulis*) supplied up to 31% of the energy demands of an associated macroinvertebrate community on the west coast of Sweden (Norling and Kautsky 2007). This process also serves as an effective water filtration purpose, with the potential of improving local water quality and clarity in areas with high-density oyster populations improving light penetration and growing conditions for submerged vegetation (Wall *et al.* 2008). In degraded ecosystems overseas where water quality is low and nutrient removal is a priority, the value of filtration by oyster communities (based on North American systems) was estimated at USD $1,385 – $6,716 ha$^{-1}$ year$^{-1}$ in 2011 (Grabowski *et al.* 2012).The impact of Australian shellfish reefs on nutrient cycling in estuaries, bays and coastal waters has not been assessed. However, a research proposal to address this lack of information has been developed, and funding is currently being sought through collaboration between The Nature Conservancy, several universities and state governments (Gillies *pers. comm.*).

### 4.4.4 Sediment stabilisation and coastal protection

Shellfish ecosystems can act as natural barriers reducing coastal erosion and protecting other habitats such as saltmarshes by reducing water velocity, baffling waves or increasing sedimentation (Meyer *et al.* 1997). Restored intertidal shellfish reefs are gaining popularity as a natural solution to control foreshore erosion and to bring complementary benefits through habitat provision and water filtration (Coen *et al.* 2007; Grabowski and Peterson 2007; Scyphers *et al.* 2011). Shellfish reefs reduce rather than deflect energy in contrast to 'grey' shoreline protection infrastructure such as seawalls, bulkheads or riprap which can further erode nearby coastal habitats. The biogenic nature of shellfish reefs means that they can repair themselves after storm damage and may be able to keep pace with sea level rise (Rodriguez *et al.* 2014). The value of oyster reefs to protect coastlines from erosion was estimated at between $US 860-86,000 ha$^{-1}$ y$^{-1}$ in 2011 values depending on the coastal environment (Grabowski *et al.* 2012). There are current research projects investigating the role of shellfish ecosystems in protecting shorelines in Australia through the University of Melbourne and by OceanWatch Australia. Shellfish reefs may have a role in reducing resuspension by armouring the substrate in soft sediment systems, but this has not been investigated in Australia.

## 4.5 Australian status of shellfish reefs

Gillies *et al.* (2018) described Australian marine shellfish ecosystems and assessed their historical and current abundance, causes for decline and past and present management. Fourteen species of bivalves were described as forming complex, three-dimensional reef or bed ecosystems in intertidal and subtidal areas across tropical, subtropical and temperate Australia. Overall, there was a lack evidence about the historical extent of shellfish ecosystems as major harvesting and documented declines began before any stock assessments (Alleway and Connell 2015; Ford and Hamer 2016; Gillies *et al.* 2018). The two most common and commercially important species, the native flat oyster *Ostrea angasi* and the native rock oyster, *Saccostrea glomerata*, have suffered dramatic declines. Currently, only one native flat oyster reef system in Georges Bay, Tasmania (Gillies *et al.* 2017) is known to exist that is comparable in size to reef systems historically harvested commercially, compared to at least 118 previously harvested locations (Gillies *et al.* 2018). Out of the 60 historically fished locations identified for native rock oysters, only six are known to still contain commercially harvestable-sized reef systems (Gillies *et al.* 2018). These are all intertidal and there are no known subtidal native rock oyster reefs left. There is evidence of similar large-scale declines in the blue mussel *Mytilus galloprovincialis*. For example, over 11,000 tonnes were commercially harvested (mostly by dredging) from Port Phillip Bay, Victoria between 1964 and 2005, but no locations with substantial beds are known by the relevant authorities in Victoria (P. Hamer *pers. comm*. in Gillies *et al.* 2018). Ecosystems developed by the introduced Pacific oyster *Crassostrea gigas* are likely to be increasing in extent, and data on the remaining ten ecosystem-forming species remains limited, preventing a detailed assessment of their status. Overall, these findings indicates that shellfish ecosystems may be among Australia's most imperilled ecosystems in the coastal environment.

Several studies have attributed the primary cause of decline in Australia to overexploitation using destructive fishing practices such as dredging and skinning (a process where all oysters and shells were raked or shovelled up and removed from intertidal oyster banks) especially during the peak years of the industry from 1850 to 1900 (Ogburn *et al.* 2007; Gillies *et al.* 2018). Oysters were harvested as food, and their shells used for production of lime to be used as a building material. Introduced predators, competitors, parasites and disease are likely to have contributed to their decline. Oysters were transported along the east coast of Australia at a large scale, and live oysters were even brought over from New Zealand, facilitating the spread of species and disease (Diggles 2013; Ogburn *et al.* 2007; Gillies *et al.* 2015; 2018). The decline of many shellfish reefs was correlated with large-scale land clearance followed by extremely large floods, this deposited huge amounts of sediment into coastal waters, which smothered bivalve ecosystems and facilitated disease (Diggles 2013). More recently, the role of contaminants and declining water quality has been identified as a cause of decline. For example, native rock oysters were functionally extinct in Sydney Harbour, most likely because of contamination from antifouling paint containing Tributyltin and recovered after Tributyltin was banned as a component (Birch *et al.* 2014). Along the east coast of Australia, many estuaries are affected by runoff from acid-sulphate soils in their catchments. When disturbed during coastal development, these sediments generate runoff that can lower the pH and cause acidification of the water which can decrease oyster survival and growth rates (Dove and Sammut 2007). Similarly, global patterns of increasing ocean acidification may challenge the recovery of shellfish ecosystems in some locations in the future (Watson *et al.* 2009, Barton *et al.* 2015; Waldbusser *et al.* 2015).

## 4.6 Shellfish reefs and Matters of National Environmental Significance

Native flat oysters (*Ostrea angasi*, <1% remaining) and rock oysters (*Saccostrea glomerata*, <8% remaining, (Gillies *et al.* 2018b) have been nominated as threatened ecosystems under the EPBC Act. A successful nomination would influence the management tools used in the conservation of these vulnerable ecosystems.

Australian shellfish reefs provide food and habitat for a range of species, and this is likely to include threatened and migratory species, including migratory bird species listed under international conventions. There has been no published research on the value of intertidal shellfish reefs (e.g. oyster banks) for migratory shorebirds in Australia but international research indicates that these are likely to provide important feeding and resting sites. For example, intertidal mussel beds have been shown to be preferred foraging habitats for many migratory birds in the Wadden Sea, an important migratory seabird habitat in Europe (Nehls *et al.* 1997; van de Kam *et al.* 1999). Invertebrates can be an important food source for migratory bird species and intertidal native rock oyster reefs on the east coast of Australia have been shown to provide habitat for macroinvertebrate communities that are five times as productive as the adjacent bare sediments (McLeod *et al. unpublished data*). It is therefore likely that the reduction in shellfish ecosystems reefs in Australia would have led to a reduction in food for migratory bird species here. It is also possible that intertidal shellfish reefs provided resting sites for a greater proportion of the tidal cycle due to their vertical structure. This is an important area for future research into the potential benefits of shellfish reef restoration.

In 2012, shellfish reefs were listed as a specific habitat type present within Ramsar wetlands, under the Convention on Wetlands of International Importance (Ramsar Convention, United Nations Educational, Scientific and Cultural Organization, 1971), which may improve the conservation and management of shellfish reefs in two ways. First, listing shellfish reefs as a critical component of Ramsar wetlands highlights the importance of these ecosystems, and second, allows new Ramsar sites to be designated based on the presence of this habitat type. Prior to this listing, the presence of shellfish reefs was not a priority for describing a Ramsar site, which has resulted in these ecosystems being historically overlooked. In their review of the 893 marine and coastal wetlands listed under the convention, Kasoar *et al.* (2015) found that almost 16% of listed sites were likely to contain shellfish reefs but failed to include shellfish in their site descriptions, while 13% provided strong or some evidence for the presence of shellfish or bivalve reefs.

This historical underreporting of bivalve reefs in connection with Ramsar sites can lead to a mismatch between baseline states recorded under site descriptions and therefore protected under the convention, and the true historical natural state of the ecosystem. For example, in a 175-page ecological character description of the Corner Inlet Ramsar wetland in Victoria there was no reference to shellfish ecosystems (BMT WBH 2011) despite these being a dominant habitat type historically (Figure 4.4).

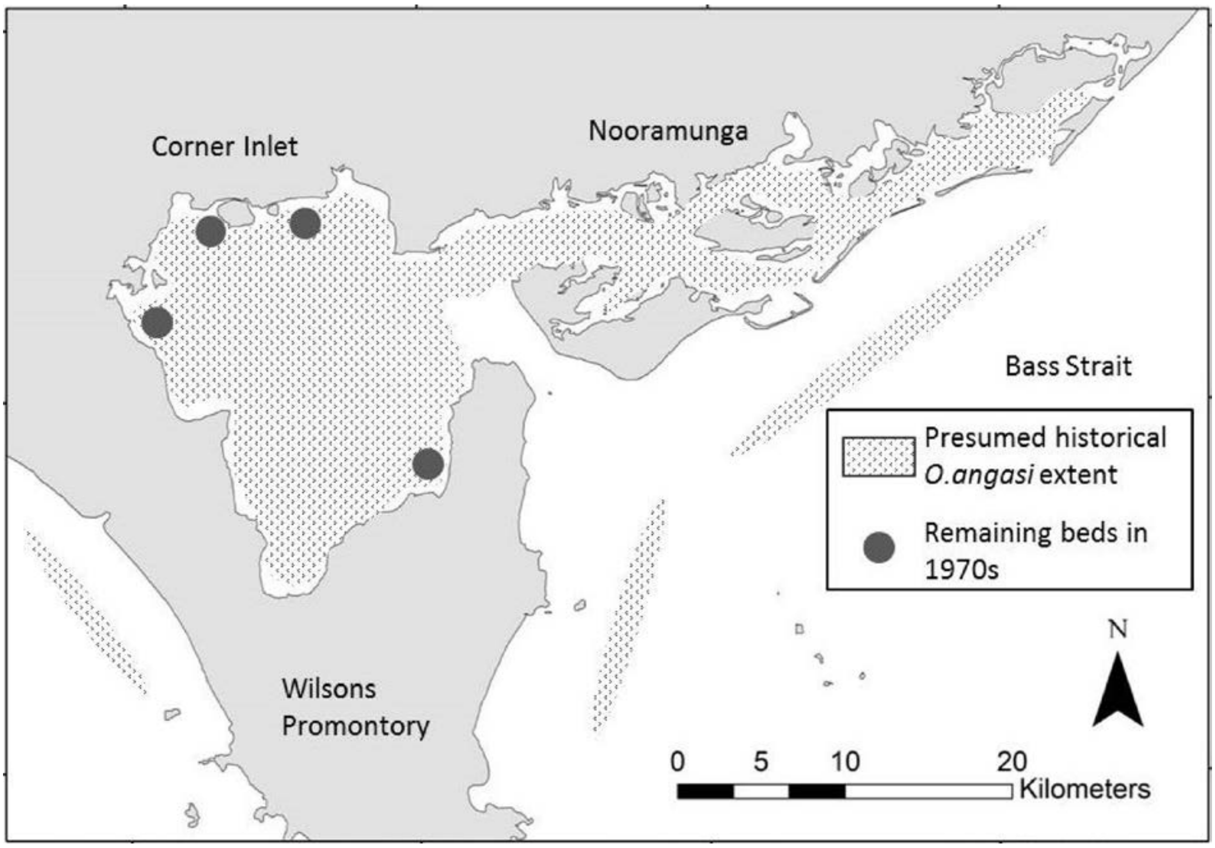


Figure 4.4: Approximate distribution of flat oysters in Corner Inlet – Nooramunga at European arrival derived from historical records and location of remaining beds in 1970s derived from fishermen's local knowledge (modified from: Ford and Hamer 2016).

## 4.7 Success and failure of shellfish reef restoration in Australia

Shellfish reef restoration is a new initiative in Australia. Restoration projects began in 2014, and now there is a growing number of projects beginning in most states. Project scale-up has been rapid with two projects already scaling up to 10s of hectares. Because most of these projects are so new, there is little information available about their success or failure, however most initial trials have showed promising results. This rapid scale-up has been possible through learning from overseas projects, particularly in the USA where oyster restoration projects have been trailed for decades, and best practice guidelines being available (e.g. Brumbaugh *et al.* 2016). The Nature Conservancy has had a lead role in facilitating the scale-up of shellfish restoration in Australia through bringing in knowledge and expertise from overseas, facilitating project progress and fundraising.

With growing interest in restoration comes an increased need for coordination and knowledge sharing. To facilitate this, the Shellfish Reef Restoration Network, supported by the The Nature Conservancy and the NESP Marine Biodiversity Hub and a range of other groups was formed in 2015 and now has over 180 members. The network is a Community of Practice, which brings together organisations and individuals interested in shellfish reef education, conservation, restoration and management. The Network aims to improve awareness of shellfish reefs and educate the broader public on the value of shellfish reef conservation and restoration. The Shellfish Reef Restoration Network also promotes communication, restoration training, policy and regulation, research and development and implementation amongst network members. Australia's growing international role in shellfish restoration was highlighted throughout the International Shellfish Restoration Conference being hosted at the University of Adelaide in February 2018, and the update of the international best practice guidelines being led by Australian practitioners and scientists.

### 4.7.1 Native flat oyster restoration

In 2014, Australia's first shellfish restoration project was initiated in Port Phillips Bay, Victoria (Gillies *et al.* 2017). Prior to European settlement, extensive areas of native flat oysters (*Ostrea angasi*) and blue mussels (*Mytilus edulis galloprovincialis*) covered up to 50% of the seafloor (Ford and Hamer 2016). These were almost eliminated through commercial dredge fishing by the mid-1990s (Ford and Hamer 2016). The project was initiated by a local recreational fishing club (The Albert Park Yachting and Angling Club), the Victorian Government (through Fisheries Victoria) and The Nature Conservancy. The project was organised in three phases. Phase one involved small-scale experimental plots where juvenile oysters were deployed onto the seafloor, with and without a base of limestone rubble. The second phase of the project (2016-18) involved deploying two ~300m$^2$ reefs comprised of a base of larger limestone rock that had juvenile native flat oysters settled onto scallop shells layered on top. The third stage of the project (2018-2021) aims to reconstruct up to 20 ha of native flat oyster reefs across the two locations (Gillies *et al.* 2017). Restoration of blue mussels is also being trialled. A shell recycling program named 'shuck don't chuck' has been initiated where used oyster, mussel and scallop shells from restaurants, venues and seafood wholesalers are collected, cured to kill off any diseases and then placed on the seafloor as a settlement substrate for juvenile oysters. Initial results have been variable but promising, with lessons learned along the way. For example, a hard substrate base provided advantages for the growth and survival of juvenile oysters (Gillies *et al.* 2017).

A similar process has been undertaken to restore native flat oyster reefs in South Australia delivered through a partnership between The Nature Conservancy, the South Australian Government, Yorke Peninsula Council and the University of Adelaide. They plan to build a $4.2 million shellfish reef across 20 ha in the Gulf of St Vincent. The first four-hectare trial reef was delivered by Primary Industries and Regions SA in 2017 and The Nature Conservancy will expand the reef by the end of 2018. There have been promising initial results with a large amount of recruitment of juvenile native flat oysters onto the built reef in 2018 (*Gillies. pers. comm.*). A third major project focussing on native flat oysters is getting underway in Oyster Harbour, WA, initiated by The Nature Conservancy in partnership with The University of Western Australia, Recfishwest and South Coast Natural Resource Management Inc (Ref). A pilot trial successfully demonstrated that native flat oysters can be collected from Oyster Harbour, spawned in the local hatchery, deployed on new reef substrate and reach the minimum level of survival required to recover a shellfish reef. Future projects plan to upscale by distributing over one million native oysters over two separate 200m$^2$ reefs in Oyster Harbour. Ultimately, the projects aims for landscape scale restoration (up to four ha of reef).

In Port River, South Australia, a citizen-led project aims to 1) restore the native flat oysters *Ostrea angasi* to the Port River; 2) provide substrate for reefs; and 3) develop reefs in areas where they can lessen the impact of storms and adverse weather events on the community. The project grew oysters on pontoons and wharves, which were deployed mid-2017 in selected restoration sites.


Figure 4.5: Oyster restoration projects in Australia.

### 4.7.2 Native rock oyster restoration

There are a number of projects beginning to trial native rock oyster restoration along the east coast of Australia. In Noosa, Queensland, the 'Bring back the fish' project funded by the Noosa Biosphere Reserve Foundation, Noosa Parks Association, Noosa Parks Association and the Thomas Foundation aims to restore oyster reefs in Noosa River In December 2017, a network of 14 patch reefs built from biodegradable bags made from coconut fibre filled with oyster shells was deployed. Project leaders hope that these will attract oyster recruits leading the development of living oyster reefs to increase fish habitat in Noosa River.

Moreton Bay historically supported some of highest densities of native rock oysters in Australia, however much like other oyster reefs around Australia, populations are greatly reduced (Diggles 2013). Local community groups have been calling for the restoration of oyster reefs to support fisheries and improve water quality in the bay. After a small pilot study, Pumicestone Passage was selected for larger scale trials for restoration of subtidal native rock oyster reefs. Three different types of engineered reef structure were deployed in December 2017 (1) a biodegradable starch matrix, (2) steel cages filled with shell and (3) patch reefs made of shell and live bivalves. Households around Bribie Island (northern Moreton Bay) were involved in the development through Australia's first community oyster gardening initiative, where community members grew bags of bivalves (native rock oysters, hairy mussels (*Trichomya hirsuta*) and rounded toothed pearl shell (*Isognomon ephippium*)) in bags hung off their pontoons in canal estates. The bivalves grown in this initiative were deployed in Pumicestone Passage as part of the trial. Partners in the project include Joondooburri Trust, Kabi Kabi First Nation, Pumicestone Passage Fish Restocking Association, Sunfish, Digsfish Services Pty Ltd, Carlo Sain, University of the Sunshine Coast, Moreton Bay Regional Council, Unitywater, Boating, Camping and Fishing, the Australian Government and the National Landcare Program, the Queensland Government, the Community Benefit Fund, Regional Landcare Facilitator Program, Bureau Waardenburg, Radboud Universiteit Nijmegen and OzFish. Preliminary results indicate that the reefs are being used visited by recreationally important fish species (B. Diggles *pers. comm.*).

### 4.7.3 Living shorelines

OceanWatch is coordinating a living shorelines project focused on shoreline protection and shellfish reef restoration. Coconut fibre mesh bags are filled with oyster shells and these are strategically pegged on eroding shorelines. In areas where the oyster spat supply is likely to be limited, the bags can be seeded prior to installation. The hope is that over time, the oysters will grow together and the coconut fibre will break down, creating a living reef. These reefs may then protect the shoreline behind from erosion and potentially accumulate sediment behind enhancing the growth of mangrove and saltmarsh community. So far trials have been deployed at nine sites across the Hastings River, Macleay River, Parramatta River and Sydney Harbour with mixed results and ongoing research into the optimal placement of the bags. Funding support for this has been provided by the Australian Government, Sydney Coastal Councils Group, Greater Sydney Local Land Services, Landcare NSW and the NSW Recreational Fishing Trust.

### 4.7.4 Enhancing abandoned aquaculture sites

There are many areas around southern and eastern Australia with abandoned oyster aquaculture leases. Many of these are located in areas that had natural reefs historically. Removing the abandoned infrastructure (e.g. piles, rocks, sticks) can be costly and damaging (K. Russell *pers. comm.*). A new initiative led by New South Wales Department of Primary Industries is attempting to rebuild native rock oyster reefs by enhancing abandoned oyster leases laying out old shell in the gaps between oyster-covered infrastructure to increase settlement substrate. This is being trailed in Port Stephens, New South Wales (K. Russell *pers. comm.*).

## 4.8 Recent advances and new ideas for shellfish reef restoration

Like many habitat restoration projects, traditional shellfish reef restoration projects have their objectives focused on recovering the primary habitat-forming species that was extirpated or degraded. Recent advancements in understanding ecosystem function, and thus ecosystem services, has helped expose a whole new range of possibilities for how habitat restoration can help address threats to coasts and estuaries and support livelihoods beyond just habitat loss. Estuary managers, communities and coastal industries can now harness shellfish reef restoration to support commercial and recreational fisheries and the bivalve aquaculture industry. Shellfish reefs are now being built to combat local pollution and eutrophication and to buffer shorelines from storm surges and sea level rise.

With the advancement and application of shellfish reef ecosystem services, innovative and long-term financing mechanisms can be established to support restoration expansion as a method to help manage and mitigate broader threats to coasts and estuaries (and associated livelihoods). For instance, the denitrification and phosphorus removal benefits derived from shellfish reefs (Newell *et al.* 2002; Kellogg *et al.* 2013; Humphries *et al.* 2016) could provide a nutrient sink mechanism with funding for restoration activities derived from estuarine nutrient trading schemes, sewerage or pollution offsets. Such programs could operate in a similar way to freshwater protection funds, which divert funding from downstream management interventions (e.g. desalination plants) to fund upper catchment restoration projects in order to secure clean water. The fisheries production benefits of shellfish reefs (zu Ermgassen *et al.* 2016) could provide a model for ecosystem-based fisheries management, whereby restoration activities are funded through recreational fisheries license funds or commercial seafood levies. The shared costs associated with developing bivalve hatcheries or research and development in bivalve genetics, disease and husbandry could be paid for in part by restoration projects, with industry cost savings returned to shellfish reef restoration projects.

## 4.9 Matters of National Environmental significance that could benefit from on-ground restoration investment

Shellfish reefs have only recently caught the attention of scientists, communities and managers in Australia. Recent studies have highlighted the historical importance of shellfish ecosystems as biogenic habitat in temperate and subtropical Australian coastal waters. This has challenged our assumptions of what is the natural state of many estuaries and coastal areas in Australia. This will have important implications for the management of MNES. For example, if baseline assessments of Ramsar wetlands were conducted in recent decades and these baselines are then used to aggressively protect that system as the status quo, this may cause a mismatch with the EPBC objectives to conserve Australian biodiversity and enhance the protection and management of important natural and cultural places as shellfish ecosystems would not be included in management plans. As restoration efforts progress in Australia, we will have important decisions to make about what should be protected. For example, if an area of seagrass in a Ramsar wetland used to be a native flat oyster reef before being removed through historical dredge fisheries, what should be protected or restored?

In section 4.2, we outline a suite of ecosystem services provided by shellfish ecosystems that may support or influence MNES and species listed under the EPBC Act (e.g. as foraging habitat for migratory birds, and their historical importance in Ramsar wetlands). This is a new area of research and many important knowledge gaps exist around their current and potential historical or restored roles. However, research on shellfish reefs overseas allows us to make some assumptions. For example, given that shellfish banks are often important feeding sites for migratory shorebirds, and we have lost the vast majority of intertidal shellfish reefs, restoration may provide feeding opportunities for migratory birds in Australia.

Native flat oyster and native rock oyster ecosystems are currently under consideration to be listed as Critically Endangered Threatened Ecological Communities under the EPBC Act. If listed these ecosystems will join giant kelp as the only marine ecosystems listed and their conservation will be a MNES. If so, this will further emphasise the importance of protecting the last remaining reference ecosystems. For example, native flat oyster reefs are functionally extinct throughout their former range across southern Australia except for one reef in Tasmania. This reef is currently subject to commercial harvest (Gillies *et al.* 2017) and listing may provide a national context to the state-based decision to continue harvest of the last reference system increasing the chance of its protection.

Australian shellfish reefs appear to be amenable to restoration, and large-scale restoration efforts have been conducted overseas, and to some extent within Australia. Shellfish reefs could therefore provide a model system to explore the role of ecological restoration in the context of protecting and preserving MNES.

## 4.10 Costs and benefits of restoration

Successful shellfish reef restoration can bring back a near extinct coastal reef system as well as the ecosystem services it provides. While the early trials around Australia are too recent to manifest substantial ecosystem recovery, there are examples from international restoration projects, which demonstrate the potential of shellfish restoration. Restoration of the eastern oyster, *Crassostrea virginica*, in North America has been demonstrated to reverse eutrophication by removing nutrients (Cerco and Noel 2007), restore fish catches (Peterson *et al.* 2003), prevent shoreline erosion (La Peyre *et al.* 2014), increase invertebrate and nekton species through habitat provisioning (Grabowski *et al.* 2005; La Peyre *et al.* 2014) and increase juvenile fish abundances.

In a recent review (Bayraktarov *et al.* 2016) evaluated the cost and feasibility of marine coastal restoration. In their meta-analysis, oyster reef restoration projects were highlighted as the second least expensive ecosystem type to restore of the five ecosystems included in the review (coral reefs, seagrass, saltmarshes, oyster reefs and mangroves, in declining order of cost). Bayraktarov et al (2016) estimated that the median restoration cost for oyster reefs is $US66,821 per hectare. Of observations that recorded survival, oyster reefs reported a median survival of 56.2%; however, the scarcity of studies that reported both survival and costs precludes an analysis of cost-effectiveness. However, given that the total value of ecosystem services provided by *Crassostrea virginica* oyster reefs has been estimated to range from US$10,325 to $99, 421 per hectare, it has been suggested that oyster reefs may recover the costs of restoration in 2-14 years (Grabowski *et al.* 2012).

The National Environmental Science Program has an ongoing project looking into the costs and benefits of the flat oyster reef restoration project in the Gulf St Vincent in South Australia. A framework for estimating the viability of shellfish reef repair projects has been developed for Australia as part of this project (Rogers *et al.* 2018).

## 4.11 Other benefits from shellfish reef restoration

Most shellfish ecosystem restoration projects are instigated with a focus on accruing ecosystem benefits such as supporting fisheries projection, improving water quality or protecting shoreline rather than focused on conservation or biodiversity outcomes.

Other beneficiaries of shellfish ecosystem restoration may include:

- Coastal communities may benefit from job creation and economic stimulus through the planning, construction works and monitoring associated with large-scale shellfish restoration projects
- Indigenous Australians through the recovery of important food sources and cultural practices and through on-Country employment opportunities, possibly through Sea-ranger programs, or Working on Country programs

Carbon burial has been touted as an incentive to restore shellfish reefs (e.g. Coen *et al.* 2007); however shellfish reefs can act as both carbon sources and sinks simultaneously. Through their filter feeding, bivalves collect particulate matter form the water column, and deposits carbon-rich biodeposits within the shell reef matrix. Through this burial, the sequestered carbon ceases to interact with faster carbon cycles, and can remain buried for hundreds or thousands of years (Ware et al. 1992; Fodrie *et al.* 2017). Meanwhile, the production of a calcium-carbonate shell binds carbon to some degree, but releases carbon dioxide and carbonic acid during the biosynthesis of the shell. Thus, shellfish reefs acts as carbon sinks during the rapid burial of carbon deposits, but as a net source of carbon during shell production, making predictions about their role as 'blue carbon' ecosystems complicated. Indeed, in a recent experiment, Fodrie et al (2017) demonstrated that the net role of restored reefs of the eastern oyster *C. virginica* as a carbon source or sink was directly related to their position on the coastal shore. Notably intertidal sandflat oyster reefs were net sources of $CO_2$ while subtidal sandflat reefs and saltmarsh-fringing oyster reefs were net carbon sinks. While the role of shellfish reefs as carbon sinks may be difficult to assess *a priori*, it is clear that the restoration of oyster reefs can facilitate the recovery of habitats that are recognised carbon sinks. For example, restored oyster reefs in North Carolina acted as natural breakwaters, dampening wave energy and increasing sediment deposition and stabilisation, allowing the surrounding saltmarsh habitat to expand seaward (Grabowski *et al.* 2005).

# 5 COASTAL SALTMARSHES

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## 5.1 Global role of coastal saltmarshes

Coastal saltmarsh in this report refers to the mosaic of coastal wetland vegetation types that occur on soft substrate shores in low energy bays, inlets and estuaries (also called tidal marshes and salt marshes). While saltmarsh ecosystems also occur in some inland areas associated with high-salinity soils, this review focuses on saltmarshes in coastal systems exclusively. Saltmarsh occurs in virtually all coastal areas globally (99 countries; Mcowen *et al.* 2017), particularly in middle to high latitudes. Coastal saltmarshes occur in the upper intertidal zone where they are regularly inundated by salt or brackish tidal waters. Due to this regular inundation, they are dominated by salt-tolerant vegetation, and support a range of infaunal and epifaunal invertebrates, as well as transient tide-dependent visitors such as fish and water birds.

Saltmarshes protect shorelines from erosion by attenuating wave energy and stabilising sediment (Morgan *et al.* 2009; Gedan and Bertness 2010; Shepard *et al.* 2011). The presence of dense vegetation, mainly grasses, reduces the velocity of passing water and decreases its turbulence, thereby reducing the erosive forces that reach the shore (Redfield 1972; Christiansen *et al.* 2000). The perforation of roots and vegetation in saltmarshes also increases the water uptake and holding capacity within saltmarshes, providing further protection from flooding and storms (Gedan and Bertness 2010; Barbier et al. 2011; Shepard *et al.* 2011). The value of this service was illustrated in a study estimating the coastal damage from 34 major US hurricanes, which found that 60% of the variation in relative damages could be explained by the presence of coastal wetlands (Costanza *et al.* 2008). The baffling effect of upright grasses in saltmarshes, which attenuates incoming wave action, also slows down outgoing water from rain, rivers and terrestrial runoff (Morgan et al 2009). The decreased speed allows sediment and particles to settle onto the benthos, effectively filtering water before it reaches the estuary (Mitsch and Gosselink 2015). Indeed, saltmarshes are so effective at cleaning water, that they have been used as substitutes for traditional municipal wastewater treatment (Richardson and Davis 1987; Breaux *et al.* 1995).

The dense and complex saltmarsh vegetation excludes larger predatory fishes, and therefore offers refuge to smaller fishes and invertebrates (Barbier *et al.* 2011). Further, the high nutrient levels provided by decaying plant matter supports accelerated growth and development of juveniles that spend part of their life cycle in the saltmarsh ecosystem (Boesch and Turner 1984). By providing habitat, nursery grounds and nutrients, saltmarshes have been demonstrated to support both recreational (Bell 1997) and commercial fisheries (Nixon 1980).

The high productivity of saltmarsh ecosystems provides an ecosystem service that may be critical in curbing future climate change - carbon sequestration (i.e. 'blue carbon'). First, the extreme productivity within the ecosystem binds carbon more effectively than terrestrial forests and many other coastal ecosystems (Figure 5.1; Macreadie et al. 2013; Mitsch and Gosselink 2015). Second, because of the slow decomposition of vegetative matter in anoxic saltmarsh soils, carbon sequestered within marshes is transferred from the short-term carbon cycle to the long-term carbon cycle (Mitsch and Gosselink 2015; Barbier *et al.* 2011). This unique shift effectively 'locks up' carbon for 1000s of years bound in peat, rather than cycling through the faster (i.e. 10-100 year) cycle common in many other ecosystems (Chmura *et al.* 2003). Together, these carbon pathways make saltmarshes (and other coastal wetlands) disproportionality important in sequestering carbon dioxide compared to terrestrial ecosystems (Mcleod *et al.* 2011).

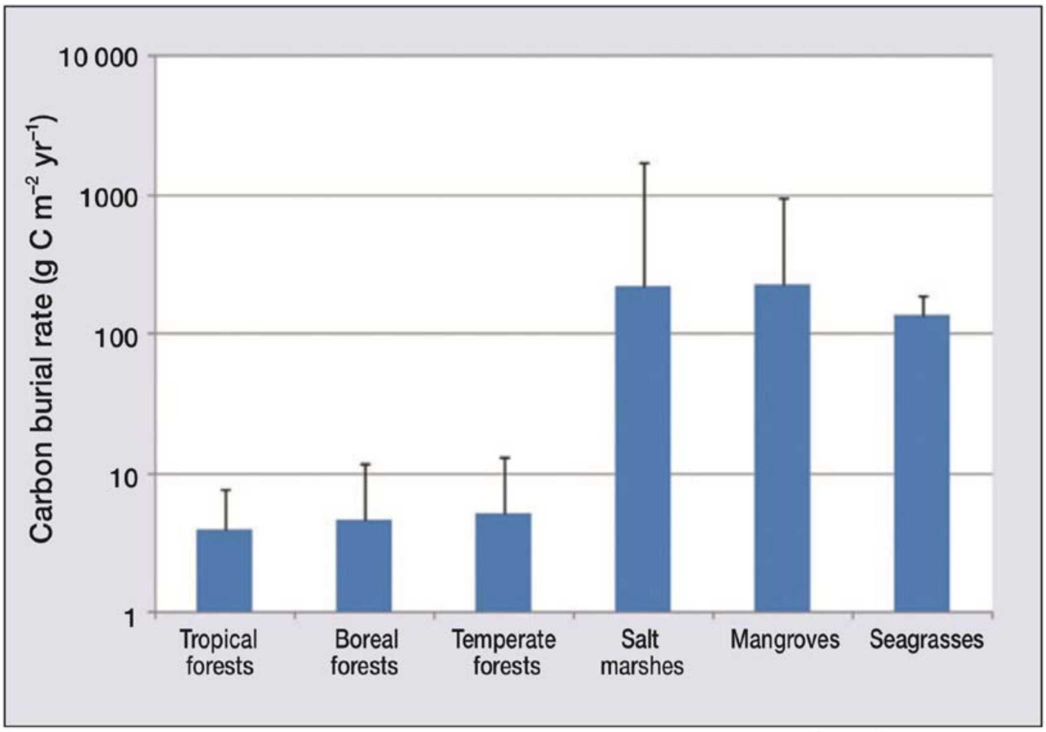


Figure 5.1: Figure 1: Mean long-term rates of C sequestration (g C $m^{-2}$ $yr^{-1}$) in soils in terrestrial forests and sediments in vegetated coastal ecosystems. Note the logarithmic scale of the y axis. (Figure from McLeod *et al.* 2011).

## 5.2 Global status of coastal saltmarshes

Their location at the interface between land and sea has made coastal saltmarshes particularly vulnerable to damage from rapidly increasing coastal populations. Until recent times saltmarshes were often viewed as useless wastelands. The most important drivers of decline include exploitation of plant production through grazing or direct harvest, drainage and reclamation for development of agriculture and building, construction of engineering works for shipping, flood protection or mosquito control, introduced species and resource extraction (van Loon-Steensma and Vellinga 2013; Doody 2007). Deterioration due to these human activities has led to a loss of 25-50% of saltmarshes worldwide (Duarte *et al.* 2008;

Barbier *et al.* 2011; Crooks *et al.* 2011). Airoldi and Beck (2011) estimate that countries in Europe have lost over 50% of their saltmarshes through coastal development. The decline of saltmarshes is not just an historical issue, but also a currently occurring problem. For example, more than 750,000 ha of coastal wetland has been reclaimed for development in China from 1985 to 2010 (Tian *et al.* 2016). This is likely to threaten the East Asian-Australasian flyway used by many Australian migratory seabirds.

New threats related to climate change include sea level rise and encroachment of mangroves into saltmarsh areas (Kelleway *et al.* 2017). Saltmarshes are naturally dynamic systems and erosion tends to be balanced by a conversion of low-lying land through flooding and the subsequent generation of new saltmarsh ecosystems (Boorman 1999). However, in recent years, coastlines have become increasingly fixed through sea walls and other engineered coastal and flood protection structures, leaving no space for naturally retreat. Some saltmarshes can keep pace with sea level rise, but in many areas, they cannot naturally retreat because of local development and in others levees and seawalls reduce sediment delivery leading to their decline (Day *et al.* 1995). Nicholls *et al.* (1999) predict that 46% of saltmarshes will be lost when sea level rises by one metre.

## 5.3 Success and failure of coastal saltmarsh restoration around the world

Given the suite of ecosystem services provided by saltmarshes, it is not surprising that a multitude of restoration projects has been undertaken around the world. These restoration projects are often part of large estuary repair projects focused on many habitat types or the whole system. Saltmarsh restoration projects can be inexpensive and small scale, for example, breaching a small bund wall to restore the flow of saline water, to extensive projects, such as the Upper St. Johns River Basin Project, in the USA, which restored 67,380 hectares of (largely freshwater) marsh areas. A successful saltmarsh restoration project built 5040 hectares of saltmarsh through weed control, altered microtopography, channel excavation and dike breaching in Delaware, USA (Weinstein *et al.* 2001). Evidence from these projects suggests that saltmarshes and the services they provide can be restored. There is a wide range of methods that can be employed to achieve this. These can be roughly categorised into five categories depending on the level of disturbance experienced by the system, and at what stage restoration is implemented: (1) removal of pressures, (2) reinstating natural hydrology (3) management of saltmarsh vegetation (4) reshaping and reprofiling, and (5) modifying sediment budgets.


Figure 5.2: Two of the most common threats against Australian saltmarshes are a) development and b) encroachment by mangroves. Photos by a) Vishnu Prahalad and b) Paul Boon.

### 5.3.1 Removal of pressures

Conserving remaining saltmarsh areas and the control or removal of the stressors causing damage to saltmarshes can be some of the most simply implemented and cost-effective restoration practices. Simple actions such as fencing to keep out large agricultural animals and control of damage from vehicles can lead to large-scale recovery.

### 5.3.2 Reinstating natural hydrology

Reversing the process of blocking natural hydrology, often by removing part of an enclosing sea wall, embankment or dyke, originally built to create new land for agriculture or other development has been used successfully in many locations around the world (Bakker *et al.* 2002; Wolters *et al.* 2005; Doody 2007). In multiple European examples (e.g. Boorman 1999; Bakker *et al.* 2002; Pethick 2002), restoration has occurred by breaching or removing seawalls, allowing seawater to flood low-lying areas and enabling saltmarsh vegetation to gradually re-establish. In some instances, saltmarsh flora was replanted, speeding up the initial process, however in one example the natural succession of plant communities took over after a few years (Boorman 1999). Changes in hydrology further allows sedimentation to occur at the inundated sites, and paired with deposits of decaying plant matter, allows the gradual build-up of the anoxic soils characteristic of saltmarshes (Boorman 1999).

### 5.3.3 Management of saltmarsh vegetation

In North America, restoration of saltmarshes has primarily been achieved through reintroduction of the dominant saltmarsh cordgrass (*Spartina alterniflora*). In one of the longest monitored restoration projects, spanning 25 years of recovery, restored saltmarshes were found to contain comparable macrophyte biomass to natural marshes as early as 5-10 years post-establishment (Craft *et al.* 1999). While the vegetation may recover relatively rapidly, epifauna and infauna may require considerably longer timeframes to recover. For example, infauna are likely to be adapted to the high carbon content characteristic of saltmarsh soils, and may therefore not recover until soil characteristics are similar to natural saltmarshes (Langis *et al.* 1991).

### 5.3.4 Modifying sediment budgets

In cases where the degraded wetland is lacking suitable elevation for the reestablishment of saltmarsh vegetation, restoration activities often involve the deposition of sediments, often through dredging. In particular, a common technique is to deposit a thin layer of sediment on top of degraded saltmarshes, to encourage rapid regrowth and recolonisation by saltmarsh vegetation (Slocum *et al.* 2005). In one such case, a thin layer of dredge spoils was sprayed on top of existing saltmarsh vegetation in Louisiana, USA. The vegetation quickly recovered from smothering, and the percent cover of saltmarsh vegetation had increased three-fold after a year (Ford *et al.* 1999). Increased vegetation biomass can effectively reverse the sediment budget from negative to positive by increasing the production of organic matter (Nyman *et al.* 1990). Subsequently, it has been theorised that thin deposits of sediment (<15cm thick) allows in situ vegetation to resprout and recolonise the treated marsh, while thicker deposits (>15 cm) smother and kill the existing vegetation, necessitating invasion by new plant material (Ford *et al.* 1999).

### 5.3.5 Controlling erosion

Novel approaches to protect saltmarsh edges include pairing revegetation restoration projects with artificial structures that attenuate waves and limit erosion, while promoting the settlement and development of ecosystem engineering species like oysters. For example, The Nature Conservancy have established several saltmarsh restoration projects that are paired with concrete structures ('oyster balls' or 'reef balls'), aimed at joint erosion control and ecosystem restoration (Gedan *et al.* 2011).

### 5.3.6 Creation of new saltmarshes

The creation of entirely new habitats is often linked to development offsets, where planned destruction of habitat is 'offset' by the creation of equivalent habitats outside the area of development. While this approach to sustainable development has been criticised for the length of time it takes for a created habitat to resemble the natural ecosystem it replaced, there is some robust evidence to suggest that saltmarsh is a particularly good candidate ecosystem. For example, there are examples in Great Britain (Atkinson et al. 2004; Morris *et al.* 2004) and the US where saltmarsh ecosystems were successfully created (Morris *et al.* 2006). However, while some vegetation may recover quickly, some evidence suggests that created saltmarshes fail to regain all the biological characteristics of the natural systems they mimic. For example, in a review of 35 created or restored saltmarshes in the UK, Mossman et al (2012) found that the species richness of restored sites rapidly mimicked that of reference sites. However, the community composition of restored sites was significantly different from reference sites, due to a dominance of early successional species with rapid growth (i.e. 'weedy species').

## 5.4 Australian role of coastal saltmarsh

Australian saltmarshes resemble those elsewhere in the world in regard to their general appearance and the general structure of the plant assemblages (Saintilan 2009). One notable difference is that these ecosystems often occur adjacent to mangroves (Saintilan and Rogers 2013) in Australia, at the upper levels of the intertidal zone where they are not subject to daily inundation, but are flooded by larger tides. Contrary to many other ecosystems, the diversity of species within the saltmarsh ecosystem increases with increasing latitudes (Adams 1996). Indeed, the southern saltmarshes have been described as some of the most diverse marsh systems on the planet (Carr 2012). While previously receiving relatively little attention, the 2010 listing of saltmarshes under the EPBC Act has generated substantial interest in the ecosystem services provided by these vulnerable ecosystems (Saintilan and Rogers 2013).

Australian coastal saltmarshes are particularly important roosting habitat for migratory shorebirds, and provide critical high-tide roosting and feeding habitat for a range of species (Prahalad *et al.* 2015; Saintilan *et al.* 2018). Indeed, almost half (18 of 42) of Australian wetlands listed as having international importance under the Ramsar convention, contain areas of saltmarsh that are considered vital for several species of migratory wading birds (Laegdsgaard 2006). Examples of migratory birds which roost and feed in Australian saltmarshes include the eastern curlew (*Numenius madagascariensis*), the Pacific golden plover (*Pluvialis fulva*), the sharp-tailed sandpiper (*Calidris acuminata*), and the red-necked stint (*Calidris ruficollis*) (Spencer *et al.* 2009). Further, saltmarsh habitats support a number of threatened species, including the orange-bellied parrot (*Neophema chrysogaster*) that feeds within saltmarsh habitats during off-breeding seasons (Loyn *et al.* 1986; Mondon *et al.*

2009), and a subspecies of yellow chat (*Epthianura crocea macgregori*) which nests and forages within central Queensland saltmarshes (Spencer *et al.* 2009). Both of these species are listed as critically endangered under the EPBC Act.

While Australian coastal saltmarsh habitats are likely to provide similar nursery grounds and adult habitat for a range of commercially and recreationally valuable fisheries species, as their international counterparts (Minello *et al.* 2003), there has been a limited number of studies which have measured this in Australia (Wegschnidal *et al.* 2017). There have been 35 fish species recorded using temperate saltmarshes (when submerged) in Australia with densities up to 72 fish per 100m$^2$ (Connolly *et al.*1997; Wegscheidl et al. 2017; Prahalad *et al.* 2018). High densities of recreationally and commercially targeted banana prawns, *Fenneropenaeus merguiensis* have been recorded within Australian saltmarshes (Connolly 2005). Other valued species that have been recorded utilising saltmarshes include flathead (*Platycephalus fuscus*), whiting (*Sillago ciliata*) and school prawns (*Metapenaeus macleayi*) (Mazumder *et al.* 2006; Connolly 2009). Saltmarshes can also indirectly support commercially valuable species by nutrient transfer through several trophic levels (i.e. a bottom-up trophic cascade). For example, saltmarsh crabs are fed on by zooplanktivorous fishes, which are prey species to commercially important fishes like the gold spot mullet (*Liza argentea*) and yellowfin bream (*Acanthopagrus australis*) (Mazumder *et al.* 2011; Saintilan and Mazumder 2017). The role of Australian saltmarshes in supporting coastal productivity has been promoted as a key justification for restoration projects (Creighton *et al.* 2017).

Australia has an estimated 13,825 km$^2$ of saltmarsh habitat, which has the capacity to sequester a vast amount of carbon. In a national comprehensive study of the sequestration capacity of Australian saltmarshes, Macreadie *et al.* (2017) estimated that they sequester 54.52 g organic carbon m$^{-2}$ yr$^{-1}$. While this is less than some saltmarshes overseas, due to differences in species composition, they still play a significant role in global and national carbon sequestration given that 33% of global saltmarsh habitat exists within Australia (Macreadie *et al.* 2017).

Australian saltmarshes provide critical protection from storms and destructive wave action. Vegetation in saltmarsh habitats attenuates wave energy and decreases shoreline erosion. Recently, interest in ‘living shorelines’, biological interventions aimed at protecting shoreline erosion, has increased in Australia and around the world (Chapman and Underwood 2011). For example, saltmarsh plants were planted in the middle of a seawall of sandstone blocks in Kogarah Bay in Sydney, helping reintroduce this lost ecosystem while still maintaining the buffering from the existing seawall (Wiecek 2009).

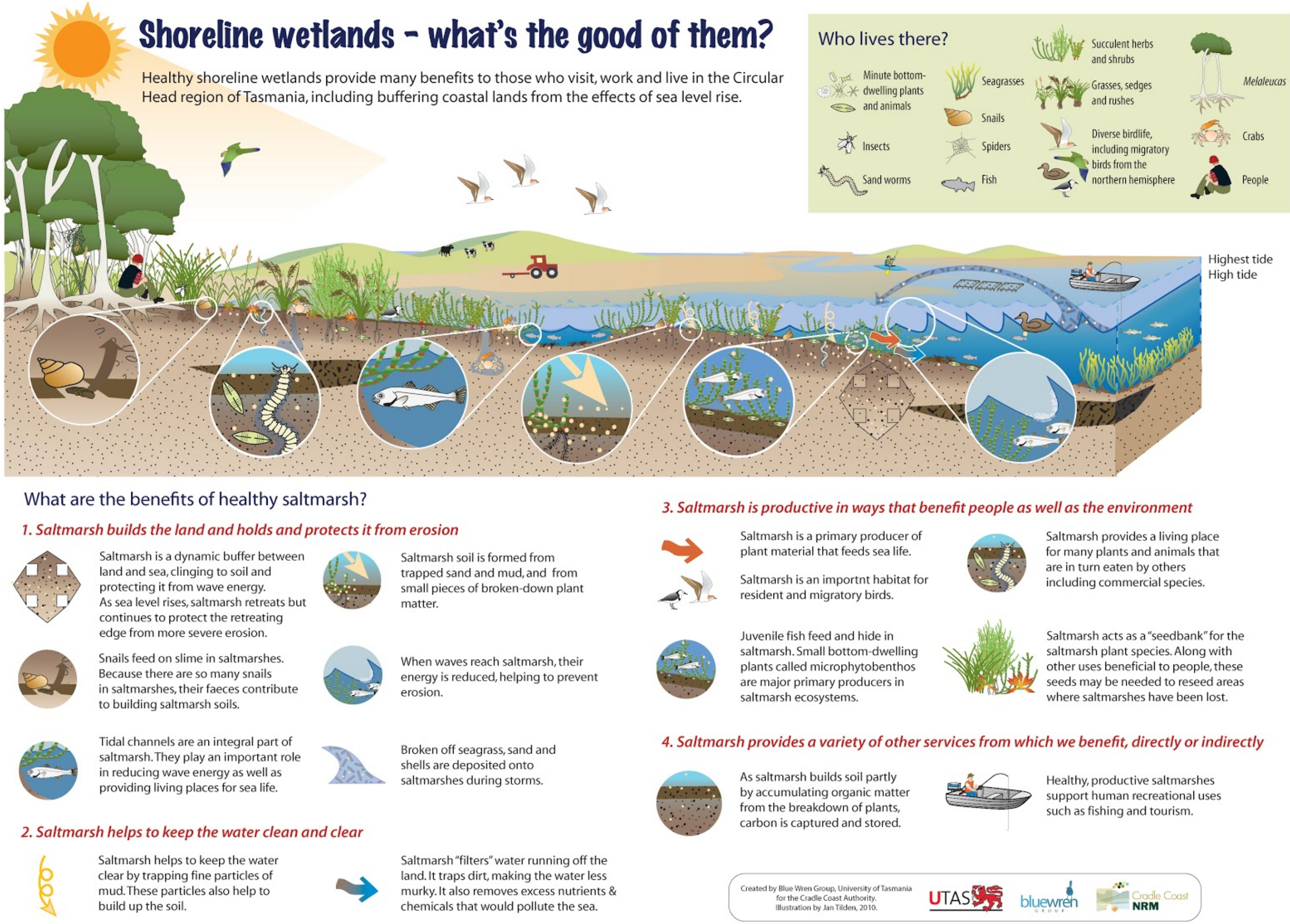


Figure 5.3: A synthesis of key coastal saltmarsh ecosystem services. Figure from Mount *et al.* (2010).

## 5.5 Australian status of coastal saltmarshes

Saltmarsh ecosystems occupied up to 16 000 km$^2$ of the Australian coastline prior to European colonisation (Saintilan 2009). Despite their important ecosystem services, these have been one of the most neglected types of wetland in Australia (Boon 2012). Throughout the 19th and 20th centuries Australian saltmarshes have been drained, filled and replaced with farmland, sports fields, houses, and canal and industrial estates (Lee *et al.* 2006; Prahalad 2014; Rogers *et al.* 2016). Indeed barriers to water flow and connectivity, such as levees, bund walls, or roads occur along almost every estuary and river in the more populated parts of Australia (Northern Land and Water Resources Australia 2002; Creighton *et al.* 2015). Other threats described in the EPBC Conservation advice for subtropical and temperate coastal saltmarshes (Threatened Species Committee 2013) are described below.

- **Clearing and fragmentation**. Historical and contemporary vegetation clearing has resulted, and will continue to result in, loss or fragmentation of coastal saltmarsh habitat. Many of the threats below cause or exacerbate this threat.
- **Altered hydrology/tidal restriction**. Changes to tidal regime or tidal connection that result from development, land-use practices or infrastructure can lead to habitat loss, invasion of 'problem species' or modification of ecological function.
- **Invasive species**. Non-native weed species and other problem species (e.g. native species that can form monotypic stands) are increasingly replacing native coastal saltmarsh plants which limits biodiversity, changes vegetation structure and potentially alters ecosystem function, and in some cases fire regimes.
- **Climate change**. Current and projected rises in temperature and sea level as well as increased storm events from climate change are considered severe threats to coastal saltmarsh that could result in landward retreat, transgression by mangroves, fragmentation and loss of habitat or function.
- **Recreation**. Various recreational vehicles cause localised and widespread damage (and noise) to coastal saltmarsh, with documented decreases and disturbance to habitat and fauna (e.g. nesting birds including ground egg layers).
- **Pollution/litter**. Pollution and litter from stormwater or dumping of waste can smother coastal saltmarsh plants and introduce contaminants such as heavy metals or organic chemicals. Oil spills are also a major potential threat.
- **Eutrophication**. Coastal saltmarsh is susceptible to a range of impacts from excess nitrogen from sewage and land-derived sources. Nitrogen can change patterns of productivity and species distribution, stimulate algal growth, and encourage non-saltmarsh vegetation to invade.
- **Acid sulfate soils**. Actual or potential acid sulfate soils are found along much of the Australian coastline and therefore pose a threat to the ecological community. Acidification can have significant impacts on habitat quality, the health of aquatic organisms and biodiversity (e.g., fish and shellfish kills, outbreaks of disease in fish, scalding of vegetation, and increases in nuisance algae).

- **Grazing**. Large-scale grazing by introduced farm animals is likely to impact on coastal saltmarsh vegetation, potentially changing composition and structure and adversely affecting rarer and more fragile species.
- **Insect control**. Controlling nuisance insects in coastal saltmarsh may involve the use of harmful insecticides or habitat modification such as runnelling, which alters drainage and tidal inundation patterns.
- **Evaporative salt production and mining**. Solar evaporative salt ponds are often constructed on coastal saltmarsh, thereby destroying vast areas of natural habitat. In South Australia, where the highest biodiversity of coastal saltmarsh occurs, vast areas are under lease for potential salt mining in the future.
- **Inappropriate fire regimes**. Coastal saltmarsh vegetation is not well fire-adapted and fire is lethal to many species. Invasive problem species (e.g. *Juncus acutus* and *Baumea juncea*) may have high flammable fuel loads, putting coastal saltmarsh at risk.

There is no comprehensive review or mapping of the loss of saltmarsh ecosystems in Australia. The rate and extent of loss has not been consistent, however, at local and regional scales, the loss of saltmarsh has been substantial. For example, it is estimated that 85% of the original saltmarsh area has been lost from Sydney Harbour (Mayer-Pinto *et al.* 2015). Saintilan and Williams (2010), reviewed loss of coastal saltmarsh in eastern Australia since World War 2, and reported 100% loss for parts of Botany Bay, New South Wales over the period 1950-1994 and 67% loss for the Hunter River (excluding Hexham) from 1954-1994. Harty and Cheng (2003) reported a loss of 78% of saltmarshes in Brisbane Water, near Gosford, New South Wales, between 1954 and 1995. Duke *et al.* (2003) described a 50% loss of saltmarsh in Moreton Bay, Queensland between 1975 and 1998. Sinclair and Boon (2012) described a 50% loss of saltmarsh and mangroves around Port Phillip Bay, Victoria. Tropical saltmarshes are extensive, but are much less studied. However, in other areas there has been minimal loss. For example in the Gulf of Carpentaria in northern Australia, there has been little loss of saltmarshes (Saintilan 2009).

Saltmarsh ecosystems face new threats through the effects of climate change. Threats include mangrove encroachment, where mangroves have moved into former saltmarsh habitat over the last few decades (Saintilan *et al.* 2014; Kelleway *et al.* 2016). For example, since 1941, 50% of saltmarsh at the Hawkesbury River mouth have been replaced by mangroves (Williams and Watford 1997). Sea level rise is also a major threat to saltmarsh if they are prevented from migrating landward by urban development of infrastructure barriers (Saintilan and Rogers 2013; Enwright *et al.* 2016).

## 5.6 Coastal saltmarshes and Matters of National Environmental Significance

Subtropical and temperate coastal saltmarsh was listed as a vulnerable ecological community under the EPBC Act in 2013. Coastal saltmarsh in the NSW North Coast, Sydney Basin and South East Corner Bioregions are also listed as endangered under the *New South Wales Threatened Species Conservation Act* 1995 (NSW) (1995). Three saltmarsh species are also listed as vulnerable under the EPBC Act, two South Australian samphire species (*Tecticornia flabelliformis* and *T. bulbosa*) and a Tasmanian leadwort species (*Limonium baudinii*).

Coastal saltmarshes provide critical habitat for listed threatened species, such as the false water rat (*Xeromys myoides*), green and golden bell frog (*Litoria aurea*), slender-billed thornbill (*Acanthiza iredalei rosinae*), Australasian bittern (*Botaurus poiciloptilus*) orange-bellied parrot (*Neophema chrysogaster*), Australian painted snipe (*Rostratula australis*) Australian fairy tern (*Sternula nereis nereis*), southern emu-wren (*Stipiturus malachurus intermedius*) and (Dawson) yellow chat (*Ephthianura crocea macgregori*). The coastal areas in and adjacent to the Great Barrier Reef World Heritage Area (GBRWHA) contain 1660 km$^2$ of saltmarsh habitat (Goudkamp and Chin 2006). The Dawson yellow chat (*Ephthianura crocea macregori*) is the most threatened bird in the GBRWHA (Goudkamp and Chin 2006). The population of the subspecies is very small and only found in saltmarsh and swampy grassland on Curtis Island and a few sites on the adjacent mainland near Gladstone (Goudkamp and Chin 2006).

Coastal saltmarshes are an important component of many RAMSAR wetlands and World Heritage Areas. Indeed, out of the 42 Australian wetlands of international importance listed under the Ramsar convention, 43% (18 of 42) contain extensive areas of saltmarsh that are considered vital for several species of migratory wading birds (Laegdsgaard 2006). A large proportion of migratory birds that travel along the East Asian-Australian flyway, utilise Australian saltmarshes for roosting or feeding during their migration (Spencer *et al.* 2009). Approximately 20% of birds on this flyway are listed as critically endangered or near threatened under IUCN risk criteria (Battley 2004). In Australia, these migratory species are listed under the EPBC Act based on international migratory bird agreements of migratory species like the Bonn Convention (Convention on the Conservation of Migratory Species of Wild Animals Appendices I and II) (United Nations Environment Program 2018), the Japan-Australia Migratory Bird Agreement (JAMBA) (Government of Japan and Government of Australia 1974), the China-Australia Migratory Bird Agreement (CAMBA) (People's Republic of China and Government of Australia 1986) or the Republic of Korea-Australia Migratory Bird Agreement (ROKAMBA) (Government of the Republic of Korea and Government of Australia 2006). Signatories to these agreements have made commitments to protect migratory bird species and the habitats, which they utilise during migrations (Spencer *et al.* 2009).

## 5.7 Success and failure of coastal saltmarsh restoration in Australia

Saltmarshes are one of the most at-risk coastal environments and current management is considered insufficient (Rogers *et al.* 2016). Subtropical and temperate coastal saltmarshes were listed as vulnerable under the EPBC Act in 2013 and the listing stated that a recovery plan and conservation advice were required. A recovery plan is yet to be published; however, the Conservation Advice identifies threats and their control and broadly suggests restoration efforts (Australian Government 2013).

Saltmarsh restoration is a relatively new activity in Australia, but restoration activities are increasing along with an increased profile and appreciation for coastal saltmarsh. Most saltmarshes are managed at a local and state level, and variation in approaches between states can make it difficult to compare management effectiveness. In addition, many restoration efforts have not been documented (Laegdsgaard 2006) meaning that we cannot learn from their success or failure. The most common saltmarsh restoration actions in Australia are fencing to remove cattle and weed removal, as well as diversion of stormwater (Laegdsgaard 2006). Grazing of cattle on saltmarshes is common practice on Australia's saltmarshes, especially in the tropical north (Anning 1980), although little evidence exists of any damage caused by grazing cattle due to the low diversity of species in low latitude saltmarshes (Saintilan*.* 2009). However, the exclusion of cattle on the heavily grazed Kooragang Island (NSW) resulted in the saltmarsh vegetation recovering in around five years (Laegdsgaard 2006).

Many restoration projects are focused on creating the right conditions for water flow to enable natural regeneration of saltmarsh communities. These projects are not often solely focused on the restoration of saltmarsh communities but on estuaries and coastal wetlands more broadly. For example, the saltmarsh community on Kooragang Island recovered relatively quickly following the removal of culverts, which restored the natural tidal flow of water in the area (Streever *et al.* 1996; Laegdsgaard 2006). Sometimes land needs to be reshaped in order to restore the correct amount of tidal inundation for saltmarshes to grow and flourish, as saltmarsh species can be sensitive to changes of a few centimetres of elevation and tidal inundation. Research into active transplantation of saltmarsh plants (cultivated or taken from donor populations) has demonstrated that they can survive and spread, although often slowly (Saintilan 2009). However, generally the best results have been achieved when the environment has been prepared for the natural recolonisation or regeneration of saltmarsh plants as plants which naturally colonise prepared areas tend to grow faster (Burchett et al. 1999a; Saintilan 2009;).

Figure 5.4: Relatively inexpensive actions such as fencing to exclude cattle scan improve the health of saltmarshes. Photo provided by Kylie Russell

Figure 5.5: Large-scale coastal wetland restoration site at Tomago, NSW. Photo by Kylie Russell.

Some restoration projects could be achieved with little effort of financial investment in on-ground works such as removing small bund walls to reinstate tidal connectivity. Other 'low hanging fruit' would include introducing culverts under roads to reinstate tidal flow that has been blocked by road development. At the larger end of the scale, a proven technique for large-scale restoration has been applied recently in NSW (Russell and Walsh 2015). This systems-wide approach includes hydrological and hydrodynamic modelling in order to fully understand the likely changes in flooding and salinity across the landscape from a range of management options, before identifying possible restoration options and associated on-ground works required. These options can then form a basis for community consultation, negotiation and works budgeting. Often remotely controlled or buoyancy controlled gates are included in the project design, so that flows can be regulated. Once works are complete, follow up monitoring of changes in vegetation, fisheries and birds throughout the restoration site is conducted and the process is adaptively managed. This process has led to successful projects at Hexham, Tomago and Big Swamp with other degraded areas being considered (Russell and Walsh 2015). These projects are currently successful at the scale of 100s of hectares are there are plans to increase the size projects to 1000s of hectares in the near future (W. Glamore, *pers. comm.*). Although no national guidelines for saltmarsh restoration have been generated, the Department of Environment and Climate Change, Government of New South Wales (2008) produced the saltwater wetlands rehabilitation manual and best practice guidelines for coastal saltmarsh, which provide a good basis for the process of developing and implementing coastal saltmarsh restoration around the country.

### 5.7.1 Case study: Indigenous-led saltmarsh restoration at Mungulla wetlands, Queensland

The Mungalla wetlands east of Ingham, on the north Queensland coast, flow into the Great Barrier Reef lagoon. They have been used as a source of food and fibre by Aboriginal Australians for thousands of years. The introduction of western agriculture around 100 years ago led to the loss and degradation of these wetlands through a combination of earth bunding (where retaining walls are used to keep salty water back) to claim land for agriculture, and pollution from upstream grazing and sugarcane agriculture, which together led to the loss of 40-90% of wetlands, with the majority of the remaining wetlands infested with weeds.

The Nywaigi Aboriginal Land Corporation acquired the land in 1999. They formed a partnership with scientific advisors from CSIRO and James Cook University to monitor and restore the ecology of the Mungalla wetlands. Simulations using sophisticated hydrodynamic modelling showed that if a bund wall was removed then during large tides, seawater would penetrate well into the wetland. The earth bund was removed in 2013 and led to a rapid ecological response. The freshwater weeds that previously infested the wetlands were immediately reduced. Within two years, the weeds were virtually absent and saltmarsh communities dominated the site. The biodiversity value of the site was vastly improved, with more aquatic species and a great abundance of birds. This approach has proven to be ecological sound and the tidal ingress will continue, cost free. A key lesson from this project is that success came from a combination of Indigenous ownership and management and scientific monitoring and management.


Figure 5.6: Photos taken 50 m above the bund wall (looking north) that show (a) the massive infestation of weeds of national significance, particularly water hyacinth before the bund was removed and (b) the enormous reduction in these weeds only two years after the bund was removed. c) Two years following the bund removal saltmarsh communities such as these native sedges, primarily *Eleocharis dulcis* (bulkuru), dominate the site. Photo by Carla Wegscheidl. d). A picture of a successful partnership. From the left; Jim Wallace (hydrologist – TropWATER); Jacob Cassady (Station Manager - Mungalla Aboriginal Corporation for Business) and Mike Nicholas (tropical weed ecologist - ex-CSIRO).

### 5.7.2 Case study: Big Swamp restoration project

Case study Author: Associate Professor William Glamore[1]

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The Big Swamp Restoration Project on the Manning River estuary in New South Wales, Australia is an example of how the dynamics of the entire estuary, along with the community needs, must be understood to successfully restore large degraded landscapes. The engineering design and assessment, in conjunction with research, planning and on-ground solutions at Big Swamp, sets a standard in wetland restoration practice. The success of this project is based on multi-disciplinary partnerships focused on hydraulics, ecology, consultation, system values, feedback loops and onsite management.

Since 2012, researchers at the Water Research Laboratory, School of Civil and Environmental Engineering at UNSW Sydney, have been working with MidCoast Council to transform over 800 hectares of Big Swamp from a large acidic landscape into an ecologically diverse wetland. The challenges faced at the Big Swamp Restoration Project site included the need to reduce production and transport of acid runoff into the Manning River estuary, encourage saltmarsh regeneration, promote endangered ecological communities and ensure flood mitigation onsite. To address these challenges, this project systematically linked numerical hydrodynamic modelling with on-ground design scenarios and an evidence-based, multi-criteria assessment method to prioritise remediation. Further, this site is one of the first large-scale tidal restoration projects to consider the implications of sea level rise and changes in salinity dynamics within on-ground remediation plans.

The practices developed in this project are now being applied at several similar sites across NSW, including Tomago Wetlands (1,000 ha) on the Hunter River estuary, Collombatti-Clybucca wetlands (3,500 ha) on the Macleay River estuary, Everlasting Swamp (3,500 ha) on the Clarence River estuary, and Tuckean Swamp (8,500 ha) on the Richmond River estuary. Further details can be found at: http://www.wrl.unsw.edu.au/research/big-swamp-restoration-project.


Fig. 5.7: Evolution of the Big Swamp Restoration Project, Before (Top) and after (Bottom) restoration (Source: ImageCatcher)

## 5.8 Recent advances and new ideas for restoring coastal saltmarshes

One of the challenges facing saltmarsh restoration projects is how to integrate adaptation planning for future climate change scenarios. Coastal saltmarshes are particularly susceptible to climate-change and sea level rise in developed coastal areas where they cannot retreat inland because of coastal development, agriculture, or infrastructure that blocks water movement. Coastal managers need to balance multiple competing uses of the coast and many aspects of coastal management are conducted at the local government level (Bradley *et al.* 2015; Clark and Johnston 2017). One forward-thinking action would be to make strategic acquisitions of land to allow for extensions of saltmarsh habitat across adaptation pathways. However, many coastal local councils, especially those with lower incomes have no, or only early stage adaptation plans (Bradley *et al.* 2015). To assist decision makers plan for climate change the National Climate Change Adaptation Research Facility developed the CoastAdapt tool, which is an information delivery and decision support framework designed to provide guidance for coastal management decisions to be made in the content of climate change and sea level rise ( National Climate Change Adaptation Research Facility 2017). One example of forward planning in coastal wetland restoration is the Hexham Swamp Rehabilitation Project, which included land acquisitions behind levees to allow for tidal inundation and saltmarsh habitats with future sea level rises (Rogers 2016).

Coastal saltmarsh has been and continues to be undervalued. A stronger focus on ecosystem valuation in an Australian context would help decision makers weigh up the relative costs and benefits of coastal, protection or restoration (Wegscheidl *et al.* 2017). However, these data are generally lacking in Australia. In a recent review, only 13 publications were found that presented quantitative information on carbon sequestration and fish production and none were found that quantified nutrient cycling, coastal protection or recreation services (Wegscheidl *et al.* 2017). A focus on benefits and functions relevant to specific groups of coastal stakeholders such as recreational and commercial fisher could encourage protection and restoration. This could open up resourcing opportunities to form public-private partnerships such as the various recreational fishing trust funds.

All coastal infrastructure has a finite lifespan so opportunities for restoration will come up when infrastructure that is built on former or potential saltmarsh habitat comes to the end of its useful lifespan. For example, the recent closure of salt field north of Adelaide has provided an opportunity for saltmarsh restoration in South Australia (Clark and Johnston 2017).

### 5.8.1 Living shoreline approaches

A living shoreline is a protected stabilised coastal edge made of natural materials such as plants, sand or rock. Unlike a concrete seawall or other hard structure, which impeded the growth of plants and animals, living shorelines grow over time. Living shorelines are growing in popularity as a cost-effective technique for shoreline protection. An advantage of living shorelines is that they provide both the physical protection and ecological function of coastal habitats. Saltmarsh plants are often incorporated into living shoreline designs or are assumed to naturally colonise an area once sediment builds up behind shellfish bank. Living shorelines can also protect saltmarshes from extreme weather events. Smith *et al.* (2018) compared living shorelines with hardened shorelines (bulkheads) in North Carolina, USA, and showed that living shorelines exhibited better resistance to landward erosion and enhanced saltmarsh growth.

### 5.8.2 Mitigating acid-sulfate soils

Many coastal wetlands in Australia including areas with extensive saltmarshes are negatively affected by acid sulfate soil that is disturbed by development. The most common treatment of acid sulfate soil is to mix an alkaline material (usually agricultural lime) into the soil so it reacts with the acidity and neutralises it. This process can be expensive and destructive at larger scales as the alkaline material needs to be physically mixed in. An innovative approach called lime-assisted tidal exchange has been trialled in Queensland. As seawater enters the wetland through a gate, it is mixed with hydrated lime to increase the amount of alkalinity and neutralise the soils when the water flows onto the land (Luke *et al.* 2017). When the tide retreats through the floodgates, its quality is monitored to make sure that the treatment is working (Luke *et al.* 2017). This techniques has proven successful at East Trinity Inlet, in Queensland as one of the world's most successful demonstrations of how to restore an areas affected by acid-sulfate soils (Luke *et al.* 2017) .

## 5.9 Matters of National Environmental Significance that could benefit from on-ground restoration investments

A recovery plan for temperate and subtropical coastal saltmarsh is required, on the recommendation of the Threatened Species Scientific Committee. However, as of August 2018, no Recovery Plan has been adopted or made for this ecological community. There is approved conservation advice for coastal saltmarshes by the DoEE including the following priority conservation actions:

- Avoid native vegetation clearance and destruction of the ecological community and its buffer zones; including protecting potential areas of natural retreat
- Collate effective policies and management actions already in progress (including development controls) to support and widely disseminate best practice and lessons learnt
- Undertake surveys to identify areas where natural retreat of coastal saltmarsh may be possible, and actively manage them to enable natural retreat in the future
- Undertake effective community engagement and education to promote the value of the ecological community (e.g. it is not 'wasteland' as some perceive); also to highlight the importance of minimising disturbance (e.g. during recreational activities), and of minimising pollution and littering (e.g. via signage)
- Liaise with planning authorities to promote the inclusion of coastal saltmarsh protection and projected tidal inundation zones in their plans/responses to climate change and sea level rise and in coastal zone management generally

Given that coastal saltmarshes are listed as threatened under the EPBC Act and endangered under the in the NSW North Coast, Sydney Basin and South East Corner Bioregions under the *New South Wales Threatened Species Conservation Act* 1995 (NSW) there could be opportunities to co-invest in restoration projects with state-based agencies. Restoration projects often avoid protected areas such as Ramsar wetlands because of perceived or actual challenges getting permits approved. This may be a lost opportunity to improve the health of protected wetlands. The Commonwealth could promote the use of restoration practices as a way to conserve and improve MNES rather than treat restoration in the same way as development projects, with similar permitting processes. Saltmarsh restoration may be an effective strategy to increase habitat and food supplies for threatened and endangered species, including migratory seabirds. For example when a population of the nationally vulnerable water mouse (*Xeromys myoides*) was identified in an intertidal wetland on the Maroochy River in Queensland, invasive weeds were controlled and mangroves were propagated to reduce erosion of coastal saltmarsh and protect the population (J. Bolzenius *pers. comm.*).

## 5.10 Estimated costs of saltmarsh restoration

The cost of saltmarsh restoration can vary substantially depending on techniques used. In a recent review of coastal restoration projects Bayraktarov *et al.* (2016) evaluated 73 projects restoring saltmarshes, and described a median cost of US$67,128 per hectare, which is comparable to the cost of oyster reef restoration, and substantially less costly than both coral reefs and seagrass restoration. However, when evaluating a subset of projects that reported both cost, area and survival the review singled out saltmarshes as the least cost-effective ecosystem for restoration compared to mangroves, seagrass, corals and oysters because saltmarsh restoration projects were only recorded in developed countries where labour costs are higher, and often included expensive engineering work to re-instate natural hydrology (Bayraktarov *et al.* 2016).

In Australia, few projects report on costs of restoration, however, the costs are likely to fall in the more expensive range outlined by Bayraktarov *et al.* (2016) given the comparatively high cost of labour and logistics in Australia. For example, 5600m$^2$ of saltmarsh at the Olympics 2000 site, Sydney, was restored through re-landscaping and revegetation, at an estimated cost of AU$80,000 per hectare, which the authors remarked was 'comparable with that of landscaping any other type of urban parkland in Sydney' (Burchett *et al.* 1999). In contrast, the Hexham Swamp Rehabilitation Project, in the Hunter River NSW, included land acquisition behind levee banks, bund construction, hydrodynamic modelling, and vegetation and wildlife surveys, and cost a total of $7 million (Rogers 2016). Give the substantial size of the restored area (650ha), this project is relatively low-cost at $AU10,769 per hectare.

## 5.11 Other benefits of coastal saltmarsh restoration

### 5.11.1 Values for Australian Traditional Owners

Wetlands have significance as ceremonial and initiation sites, traditional hunting and gathering grounds such as boundary markers. Almost all wetland plants and animals have some form of traditional use as food, fibre, containers, tools, weapons, transport, shelter and medicine.

### 5.11.2 Blue carbon

While Australian saltmarshes harbour different species, and sequester less carbon than their overseas counterparts (54.52 g organic carbon $m^2$ $yr^{-1}$), they still play a significant role in global and national carbon sequestration due to their large extent with one-third of the world's saltmarshes occurring in Australia (Macreadie *et al.* 2017). High levels of productivity combined with long retention times makes saltmarshes effective carbon sinks (Macreadie et al 2017), and the link to restoring this critical ecosystem service is clear (Connor *et al.* 2001). While Australian saltmarshes may sequester lower amounts of carbon, due to differences in vegetation community compared to international counterparts, they still sequester several times the amount sequestered be terrestrial systems and thus play an important part in reducing carbon in the atmosphere. Successful restoration of saltmarshes may therefore increase the blue carbon potential in Australian coastal wetlands. An important exception is in the case of mangrove encroachment into saltmarshes, where the invading ecosystem (mangrove) actually sequester more carbon than the saltmarshes themselves (Kelleway *et al.* 2016).

# 6 MARINE AND COASTAL HABITAT RESTORATION WORKSHOP IN CANBERRA, JUNE 2018

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The Marine and Coastal Habitat Restoration symposium was held over two days from June 20-21, 2018 at the Department of the Environment and Energy (DoEE) in Canberra. Similarly, to previous symposia (e.g. the 2017 Australian Coastal Restoration Symposium) the symposium was informal, with an 'invitation-only' format. It followed a programme where presentations were given under two main themes; (1) Setting the scene of marine and coastal restoration in Australia, and (2) Enabling restoration. In the morning session, we asked the question 'How can restoration complement existing management of coastal habitats in a changing world?' In the afternoon session we asked 'What is needed to enable restoration?' and 'What are the best mechanisms for facilitating national coordination of research and funding?' In between sessions, there was time for networking conversations during lunch and tea breaks.

Both sessions offered opportunities to a wide range of restoration professionals to give 10-minute talks. A keynote was delivered by Dr Chris Gilles from The Nature Conservancy, who discussed scaling up coastal habitat restoration in Australia. Delegates at this symposium included representatives from the Department of the Environment and Energy, Department of Primary Industries and Regions (SA), Department of Foreign Affairs and Trade, Australian Institute of Marine Science, Sydney Institute of Marine Science, The Nature Conservancy, Shellfish Reef Restoration Network, Seagrass Restoration Network, Mangrove and Saltmarsh Network, OceanWatch, OzFish, TropWATER, NESP Tropical Water Quality Hub, NESP Marine Biodiversity Hub, CoastAdapt, Reef and Rainforest Research Centre, Fisheries Research and Development Corporation, Community Seagrass Restoration, and Universities of Adelaide, Deakin, James Cook, Macquarie, Melbourne, Murdoch, New South Wales, Queensland, Tasmania and Western Australia.

Representatives from the DoEE were from the following sections:

- State of the Environment Section
- Partnerships Section
- Marine and Freshwater Species Conservation Section
- Environmental Standards Division, NSW Assessments South Section
- Wetlands Section
- Migratory Species Section
- Natural Heritage Section

- Reef Trust Section
- Environmental Standards Division, QLD South and Sea Dumping Section
- Environmental Standards Division, Major Projects West Section
- Climate Change Division, Agriculture Section
- Emission Reduction Fund Governance and Policy Section
- International Climate Change and Energy Division, Land Inventory Section
- Mitigation and Climate Science Section

Separate to the speaking sessions, a listening session allowed the Department of the Environment and Energy to engage directly with marine coastal restoration academics and practitioners in an open environment. The DoEE have designed a co-design process, with an aim to develop capability and capacity for environmental outcomes. As part of this process, the DoEE offered to facilitate conversations between parties and share contacts. A panel discussion, facilitated by Paul Hedge was inserted between speaking sessions, to open the floor to any responses to the symposium objectives. The following discussion points were explored:

- Is there a need for national coordination of research and funding for restoration and what are the best mechanisms for this?
- What Federal Departments should be involved?
- What is the best group at DoEE to be a contact point for restoration projects?
- Should restoration be focused on or excluded from protected areas - e.g. World Heritage Areas, RAMSAR wetlands
- Who should pay for restoration?
- How does restoration fit within the context of resilience-based management and a changing climate?

As a result of the panel discussions, there was general agreement that there was a need to (1) agree on a shared vision within the national marine and coastal restoration community at a broader level than habitat types and states, (2) effectively communicate this vision to the public, stakeholders and beneficiaries via a (3) national network to encourage collaboration with business, industry, government, academia, and the public with partnerships based on (4) shared value across sectors.

## 6.1 The Department of the Environment and Energy co-design process

Since December 2017, the DoEE has worked with over 149 people and more than 100 organisations on a co-design process to explore partnering for environmental outcomes. As a result, the DoEE support development of new opportunities to engage with partners and collaborate with a range of different sectors for improved environmental outcomes. To initiate this, the DoEE have implemented a co-design process to explore partnering for environmental outcomes, the first phase of which is focused on communication. They propose to be a central point of contact to connect potential partners and build capacity for more cross-sector partnerships. Through the co-design process, sectors have been identified (e.g. business, academia, civil society and government) for the value they bring to the table in terms of partnerships and what role and expertise can be offered. The aim of this is to align public and private investment, promote innovation, and enable effective joint efforts for environmental outcomes. The DoEE sees the role of the Australian Coastal Restoration Network and academia as a trusted source of information, global networks, knowledge production and distribution, with access to educating the next generation of students.

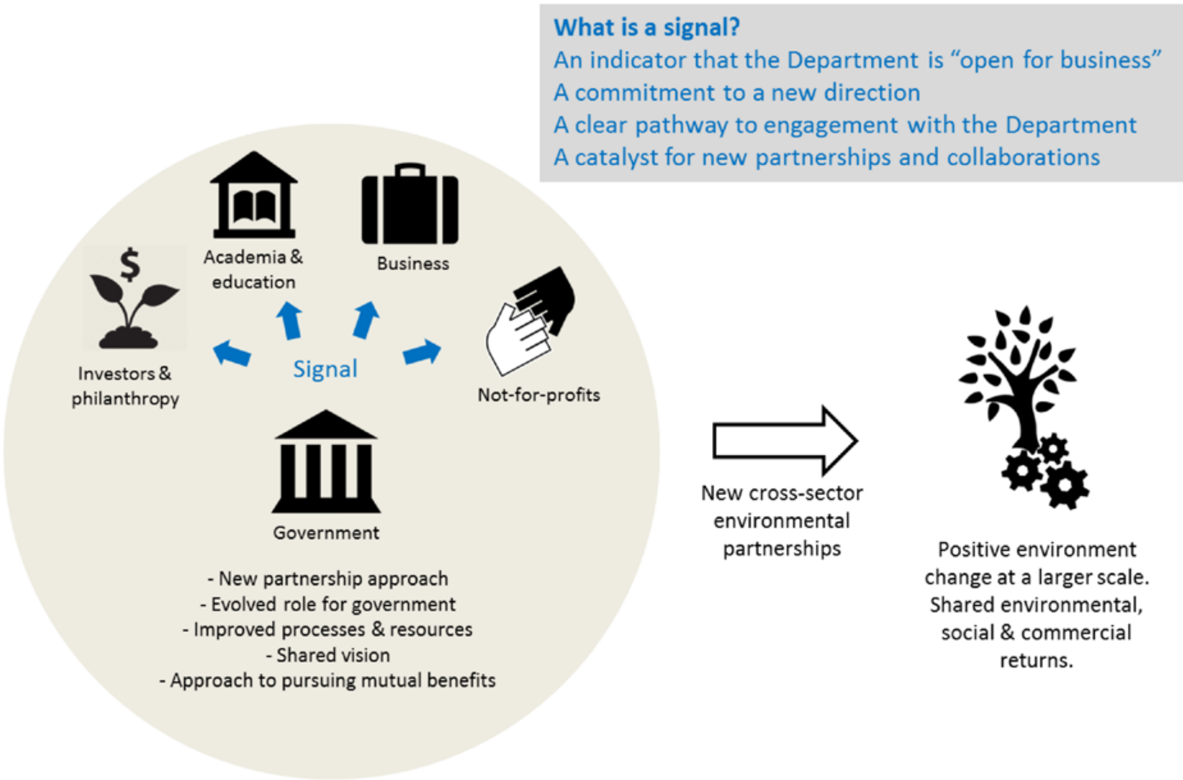


Figure 6.1 The DoEE Co-Design process.

### 6.1.1 Shared vision

There was a consensus amongst attendees that it would be useful to think about and discuss a national shared vision or overarching goal for marine coastal restoration that transcended habitat types. Questions such as 'how should we prioritise restoration?' and 'what scope and scale of restoration should we aim for?' were asked. It was also noted that downstream benefits of restoration to other sectors such as fisheries, water quality, ecosystem services etc. should be clearly delineated for all restoration projects. Understanding and communication of the broader objectives of restoration were highlighted. An example where a shared vision brought different stakeholders together was after the mass bleaching events in the Great Barrier Reef (GBR) in 2016 and 2017 (Damien Burrows, *pers. comm.*). The NESP Tropical Water Quality Hub has previously focused on restoring water quality, which had benefited the agricultural industry. After the bleaching events, the agriculture industry was supportive of NESP to switch focus to restoration, despite a perceived loss of benefits. This flexibility and adaptive style of management was made possible by a shared vision between stakeholders for a healthy GBR. Shared visions and goals may help marine restoration to avoid the tragedy of the commons, where a lack of ownership leads to overexploitation of marine resources.

### 6.1.2 Communication

A point that came up repeatedly in the panel discussion was that communication of restoration outcomes needs to be more effective. Practitioners need to ensure that they have articulated benefits to beneficiaries including the public and other stakeholders. Representative from the Fisheries Research and Development Corporation and OzFish in particular noted the importance of the fishery industry as key stakeholders with whom the restoration community have shared value (for more on shared value see below). A suggestion was made to capture the public's attention and to encourage public advocacy of restoration projects by prioritising restoration in populated areas (e.g. Port Phillip, Melbourne). This strategy aims to communicate directly with the public and show them the benefits in a way that is easily seen. Communication of restoration in this way is a political and social activity that will hopefully lead to funding and restoration success.

### 6.1.3 Collaboration

There was general support for the idea of a national level restoration network to encourage collaboration. Collaboration was endorsed as opposed to coordination to allow for diversity of restoration approaches working towards a common goal (see shared vision above). A national network should work towards mentoring each other and sharing ideas, mobilising a range of techniques, and identifying key areas to improve. Projects would be connected not by shared restoration techniques but instead by a broad understanding of where you want to go, and you take everyone with you of course (Lowri Price, *Pers. Comm.*).

### 6.1.4 Shared value

Collaboration should extend beyond the realms of ecology, and identifying shared value with the private sector will be a valuable tool to increase funding to restoration projects. In this way, downstream value can be communicated to stakeholders in order to fund upstream restoration. Once shared value has been identified, we need to articulate the benefits and bring various sectors together (see collaboration above). Examples of sectors where shared value can be used to produce mutual benefits include fisheries (wild caught, aquaculture, and recreational), agriculture and tourism sectors. This model has potential to move away from smaller funding in the tens of thousands and instead towards millions, which could fund systems-wide restoration at landscape scales.

# 7 CASE STUDY: RESTORATION ECOLOGY FOR SPOTTED HANDFISH

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Once a locally common fish of South East Tasmania, the spotted handfish (*Brachionichthys hirsutus*) has suffered a severe decline in numbers. This collapse in the global population was first noticed in the early 1990s, and resulted in its declaration as the first marine fish listed as critically endangered on the IUCN Red List (Bruce and Last, 1996) and the subsequent commencement of a series of recovery plans. Recent work has identified that handfish are a habitat specialist (Wong *et al.* 2018), choosing to live and spawn in shallow, sheltered water bays with biologically complex sea floors. In addition, spotted handfish also have an unusual symbiotic relationship with soft sediment invertebrates, where fish will spawn their eggmass around organisms like stalked ascidian. Unfortunately, this habitat complexity and breeding substrates are being destroyed by an introduced marine pest, the North Pacific sea star, a voracious generalist predator, as well as by mechanical disturbance from the chain of swing moorings that anchor yachts (Figure 7.1a, b). Like the handfish, both sea stars and moorings are also concentrated in the shallow bays of South East Tasmania. To counter these on-going threatening processes CSIRO, NESP, UTAS and DoEE are undertaking two ecological restoration projects. The first is to plant 5000 artificial spawning habitats (Figure 7.1b, c) across five known population hotspots for the fish, while the second is to replace swing moorings with environmentally sensitive gear. Artificial spawning habitats have a long history as an effective conservation measures for the handfish, as previous plastic versions are inedible to the starfish and known to be used for breeding (Figure 7.1b, d). Over the last 18 years these have been planted at multiple sites, working a little like ‘nesting boxes’ for rare birds, and may have contributed to an observed stabilisation of the decline in the species in recent years. Over time, these light plastic versions are eventually knocked over by bio-fouling, the movements of large snails or as skates and rays worked the seafloor. In our current project, new ceramic artificial spawning habitats are being trialled to replace the well-established plastic versions in the hope that they will be a longer lasting solution. For the environmentally sensitive moorings, we are also approaching the problem slightly differently than previous attempts. By working with citizen scientists and industry, rather than directly with government, we want to not only observe if environmentally sensitive moorings restore habitat but also how to encourage mass adoption. Hence, besides deploying environmentally sensitive moorings and monitoring the biological effects we are also undertaking engineering modelling to check the integrity of the design and perception studies to discover any barriers to broader uptake by the boating community.

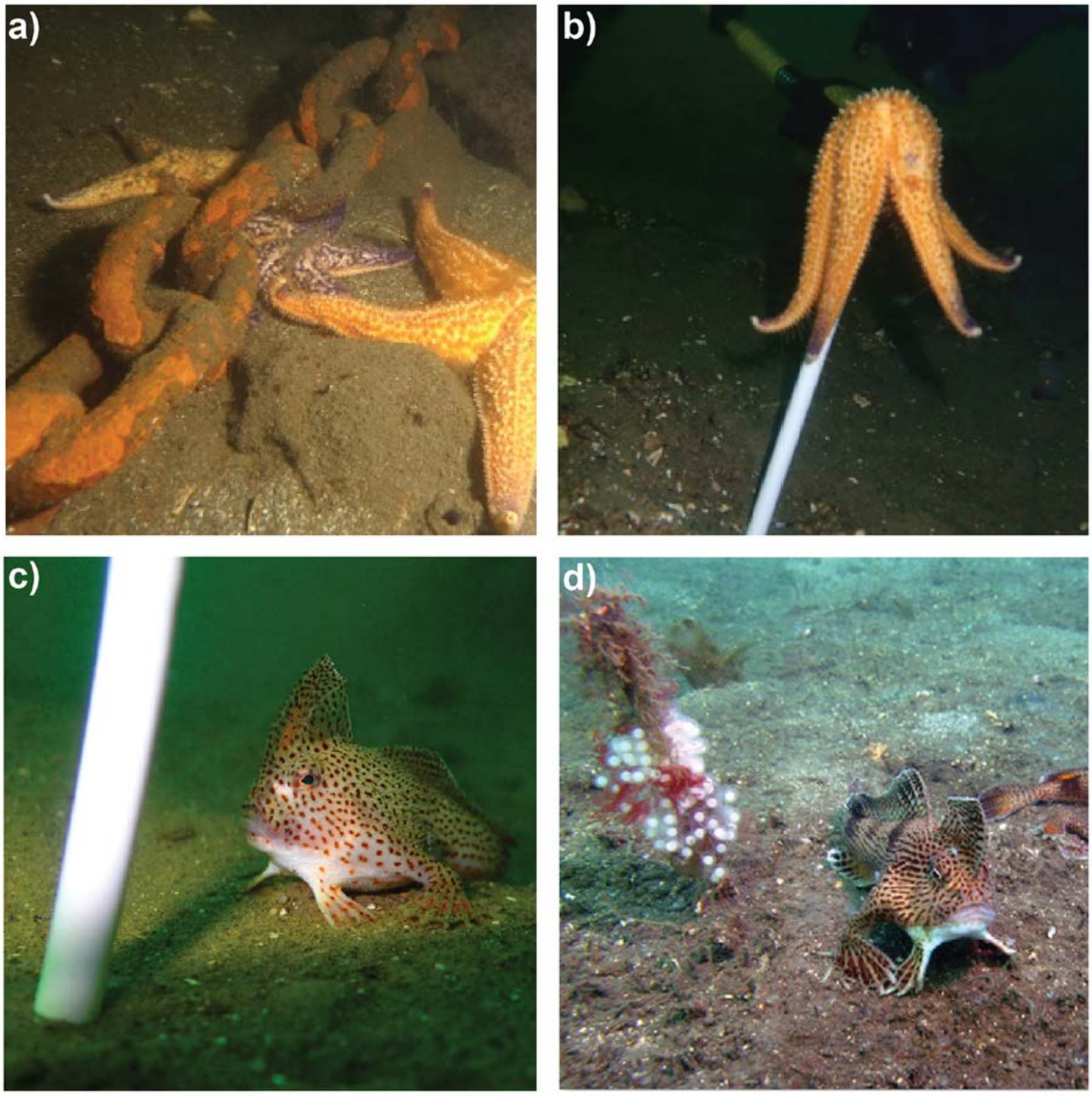


Figure 7.1: The habitat for the spotted handfish is threatened by swing moorings and the Northern Pacific seastar (a, b). To help the species recover, researchers at CSIRO have developed an artificial spawning habitat (b, c), onto which the handfish deposits eggs. The handfish stay near the eggs (c) until they hatch as fully formed juveniles, with no intermediate planktonic phase. Photo credits: a) Tim Lynch, b, c) Laura Smith, d) Antonia Cooper.

# 8 DISCUSSION

The four case studies have highlighted substantial losses of important marine and coastal habitats and the ecosystems services they provide. In the context of chronic stressors and new challenges such as climate change it appears these habitats are likely to continue to decline. Much of the management responsibility for these habitats lies with relevant state, territory and local agencies. However, each of the four habitat types described also fall under Commonwealth responsibilities through the EPBC Act. Further, these responsibilities may increase as more habitats and species are listed.

Australia is a recognised global leader in marine research, national resource management and national landcare initiatives (Gillies *et al.* 2015). Interventions such as tree planting and pest control are accepted management practices in terrestrial systems, as are enhancing fish passage and connectivity in river systems. However, until recently marine and coastal habitat restoration has not been a commonly used management tool. Other countries such as the US, Canada and the UK have embraced the need for restoration at a large scale (Gillies *et al.* 2015). For example, restoration was estimated to contribute almost US$25 billion and 221,000 jobs annually to the US economy (BenDor *et al.* 2015). Following this trend demand for marine and coastal restoration is increasing rapidly in Australia and restoration activities are scaling up rapidly, especially projects focused on coastal wetlands or rebuilding shellfish reefs.

So what role does the Commonwealth have within this changing landscape? Australia has an opportunity to avoid the common growing pains associated with developing the restoration economy and industry, by learning from mature projects and policies overseas. The federal government can show strong leadership for state, territory and local managers by taking a coordination role and providing guidance to encourage best practice ecological restoration. The Commonwealth also has responsibility for each of the four habitat types described in this report in some areas through the EPBC Act and MNES.

## 8.1 Does restoration work?

Given that ecological restoration is a relatively new endeavour in many ecosystems across Australia, success has overall been patchy. Further, while some pilot projects have yielded successful outcomes, there are still issues with scaling up in many systems. However, with decades of experience from overseas to draw on, and with recent advances in restoration ecology, Australia has an opportunity to avoid the common growing pains experienced elsewhere. Scientific reviews necessarily judge the feasibility of restoration based on historical success rates (often over a decadal timescale). However, recent success and innovative approaches offer hope for increased success rates in the future, assuming that we measure and report success and failure, adaptively manage projects, and share knowledge to build capacity.

A common theme amongst the four ecosystems reviewed in this report is the variability of outcomes from restoration projects. For example, despite decades of restoration practice on Australian seagrass species, our ability to improve the survivorship of outplants still remains relatively variable. While some projects have been highly successful (i.e. >90% survival), the majority of projects still report outplant survival at less than 25%. It is however important to note that a high proportion of these (~70%, Appendix II) were research focused (e.g. testing ecological theories, techniques, or locations) rather than commercial-scale restoration attempts. This skew in objectives will clearly have an effect when comparing the size of areas re-planted, scalability, duration of success and monitoring. However, building on decades of attempts in the US, successful scale-up seems possible, and even necessary to ensure return on investment (Hernández *et al.* 2018; Reynolds *et al.* 2016). Ongoing research through the Seagrass Restoration Network will provide a framework for how seagrass restoration fits within resilience-based management and a toolbox of restoration techniques suitable for Australian conditions (G. Kendrick *pers. comm.*).

To the best of our knowledge, there are only two examples of attempted kelp forest restoration in Australia. The first, describes attempts to restore giant kelp (*Macrocystis pyrifera*) forests at several locations in Tasmania (Sanderson 2003). The outcomes of the project was limited, with short-term success achieved at just one site. The second example is Operation Crayweed, which began in 2012 and aims to restore crayweed (*Phyllospora comosa*) forests to metropolitan Sydney where they were once abundant (Campbell *et al.* 2014; Marzinelli *et al.* 2014, 2016, *unpublished data*; OperationCrayweed.com). Outplanted crayweed patches have rapidly become self-sustaining with no additional cost or maintenance, which is a rare result in marine restoration. Critically, this relatively small-scale intervention appears to have translated into a large-scale impact, with crayweed populations continuing to expand and colonise substantial areas and beginning to function as natural forests (Marzinelli *et al.* 2016, Marzinelli *et al. unpublished data*)

Shellfish reef restoration is a new endeavour in Australia with the first trial projects starting in 2014, along with an increase in related research into the function and structure of shellfish reefs. Two projects focused on building native flat oyster reefs in South Australia and Victoria have scaled up rapidly to 10s of hectares, with plans to build 100s of hectares of reefs in the near future. This rapid scaling up has relied on using techniques shown to work in the US as well as leadership and coinvestment from The Nature Conservancy. Initial results are promising, with adult bivalves surviving, bivalves recruiting to built reefs and targeted fish species associating with newly built reefs. These initial results are encouraging and broadly important in the content of conservation and endangered species management, as these reefs were functionally extinct in both of these states (Gillies *et al*. 2018). For native rock oysters, projects focused on increasing substrate within degraded intertidal oyster reefs seem likely to succeed as any structure in the areas are often rapidly colonised by oysters. Sub-tidal native rock oyster restoration may be more challenging, as subtidal native rock oyster reefs are also functionally extinct, despite intertidal reefs providing a potential source of recruits. Disease outbreaks have plagued Australia’s bivalve aquaculture industry and are likely to threaten shellfish reef restoration projects. Strategies to mitigate this threat have been built into many projects.

Of the four habitats described in this report, saltmarsh restoration as a component of coastal wetland restoration appears to be the most advanced. Success has been described at the scale of 100s of hectares, with plans to expand projects to 1000s of hectares in the near future in NSW. The approach outlined for projects in NSW that includes hydrological modelling, wide stakeholder engagement and strategic land acquisition could be used as a model of success for other regions in Australia. These projects rely on returning natural flows of seawater to facilitate the growth of saltmarshes and other habitats. However, these systems will need ongoing maintenance and will continue to be managed systems into the future, as gates and other built infrastructure to control flow for risk management are critical components of their design.

Restoration has been shown to work in some circumstances, but more experience and research is needed to see if restoration can be scaled-up and techniques modified to be effective for other ecosystem types. There have been a many ineffective restoration projects. Therefore, restoration has the potential to be a valuable tool for marine and coastal habitats but each project will need to carefully evaluated and the results monitored so we can continue to refine when and how restoration can be used effectively.

## 8.2 When is restoration an appropriate management tool?

The Australasian chapter of Society for Ecological Restoration (SERA), the world's leading ecological restoration body, has developed the 'National Restoration Standards' (Standards Reference Group SERA 2017), which outlines the guiding principles and minimum standards expected of an ecological restoration project, placed in the context of Australia's unique biodiversity and ecological complexity. In this guiding document, SERA highlights the conditions where ecological restoration should be considered. Depending on the level of degradation, some systems may recover naturally if the stressor is removed. An example of this is the recovery of intertidal native rock oysters in Sydney Harbour, which recovered naturally once a toxin included in antifouling paint was, banned (Birch *et al.* 2014).

However, systems that are compromised enough that a biotic barrier stops (or dramatically slows down) the natural recovery of the system, even when the primary stressor is removed, may require interventions to assist its recovery (Johnson *et al.* 2017; Figure 3.1). Figure 3.1 is an example of a decision making tool to determine when restoration is or is not appropriate. A case study that highlights this scenario is Operation Crayweed. Crayweed (*Phyllospora comosa*) once formed dense beds on shallow reefs along the Sydney coastline. This habitat-forming seaweed disappeared in the 1980s, presumably due to declining coastal water quality from poorly treated sewage outflows. Despite almost three decades of dramatically improved water quality in the bay, the crayweed failed to recover naturally (Coleman *et al.* 2008). However, by transplanting healthy fertile adults in degraded areas, patches of restored crayweed have begun to naturally expand and reproduce, and are supporting a diverse range of epifaunal organisms (Marzinelli *et al.* 2016; Campbell *et al.* 2014).

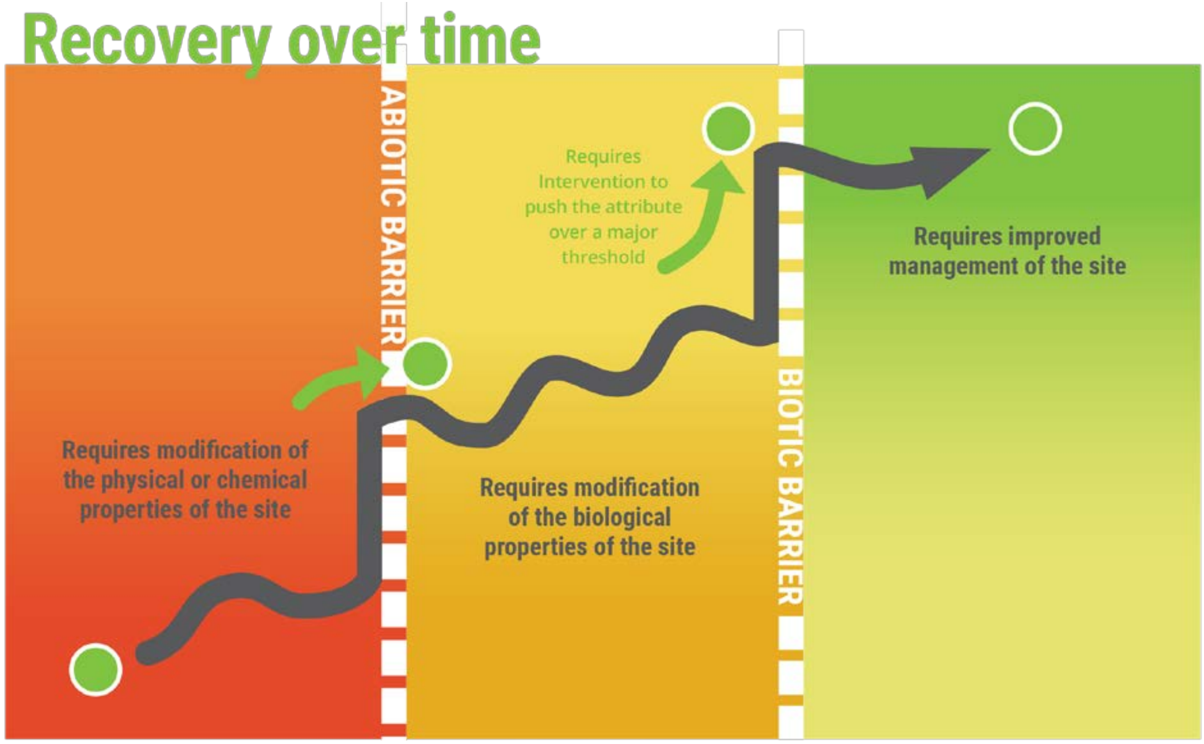


Figure 8.1: The course of recovery does not always run smoothly, and restoration interventions are often needed to 'push' an ecosystem towards a higher level of recovery. In a severely degraded site, (orange panel) intensive physical interventions may be needed. In sites where degradation is more moderate (yellow panel), recovery can be initiated by intermediate-level interventions such as introducing desired species and removing undesired species. In sites where degradation is low (green panel), recovery may be achieved through activities that prevent any further degradation of the site (e.g. fencing to keep out stock) or that reinstate important ecosystem functions (e.g. reinstating flooding or fire regimes to encourage the return of desirable species). Restoration planners and practitioners must carefully assess what is required before implementing a treatment. At some sites, physical interventions may be all that is needed to encourage plant and animal colonisation. Figure and caption used with permission from SERA, based on SER National Restoration Standards.

Similarly, subtidal Sydney rock oysters (*Saccostrea glomerata*) were once abundant over vast portions of Australia's coastlines, but were extensively harvested in the late 1800's until no extant populations of subtidal oyster reefs remain today (Gillies *et al.* 2018). However, despite the cessation of harvest, these ecosystems have failed to recover throughout their historical range, suggesting that an intervention activity may be required to encourage natural recovery.

Substantial barriers that have prevented restoration from being a widely used tool for conservation include the considerable cost, difficulties with scaling up and the significant knowledge gaps that still exist on best practice ecological restoration for coastal ecosystems. In a recent review, Bayraktarov *et al.* (2016) analysed the global cost and feasibility of restoration of three of the ecosystems outlined in this review (seagrass, oysters and saltmarshes), as well as two ecosystems outside the scope of this review (coral reefs and mangroves). They highlighted the costs associated with restoration in each of the systems, ranging from around US$67,000 per hectare for oysters and saltmarshes to over a US$100,000 per hectare for seagrasses. Further, they did not find evidence for economy of scale, where cost per hectare is reduced for larger projects; however, they speculate that this may be due to a paucity of larger projects in the data set. It should be noted however that the data on costs in each ecosystem is heavily influenced by the drivers of restoration, so that restoration projects that typically draw community and volunteer participation (like shellfish restoration) are substantially less costly than those primarily driven through environmental mitigation schemes (i.e. offsets) from industry.

Despite these challenges, there is a growing recognition that the protectionist approach (i.e. protect threatened communities and relying on natural recovery only) as a means to preserve Australia's coastal ecosystems and their ecosystem services may no longer be sufficient. It has been argued that active interventions to promote the recovery and resilience of many marine ecosystems are now required (e.g. Aronson and Alexander 2013; Gillies *et al.* 2015; Possingham *et al.* 2015; Anthony *et al.* 2017).

## 8.3 What is the role of restoration for the conservation of MNES?

Historically, conservation has focused on reducing stressors such as damage from development; however, conservation as a whole is becoming more interventionist (Hobbs *et al.* 2011). As the threats facing biological communities around the globe are expanding and compounding, it is becoming increasingly clear that habitat protection by itself is not sufficient to curb the loss of biodiversity, critical habitats and ecosystem services (Marvier *et al*. 2011). Within the context of Australia's MNES restoration could serve as a valuable tool in the toolkit to preserve vulnerable ecosystems. Given the current biodiversity declines in the shallow coastal marine habitats, it seems prudent to consider all possible management actions, including interventions.

An instrument through which to regulate and encourage restoration could be through the recovery plans of a threatened species or ecosystems. Recovery plans set out the research and management actions necessary to stop the decline of, and support the recovery of, listed threatened species or threatened ecological communities. The aim of a recovery plan is to maximise the long-term survival in the wild of a threatened species or ecological community (EPBC Act, 1995). Given the role of recovery plans as tools to describe a path to recovery for ecosystems listed under the EPBC Act, this would be an appropriate forum for restoration advice and guidance.

When considering restoration of MNES it will be critical to evaluate the current threats facing a species or habitat, and identify critical life stages or habitats that may be the barriers that prohibit natural recovery. Restoration activities should be tailored to, and focused on addressing these barriers. An example of this type of targeted approach can be found in the Raine Island Recovery Project, which was aimed at protecting the green turtle population for which the island is an important nesting site. Beach erosion has caused tidal inundation of nesting sites, drowning newly laid eggs and reducing hatching success. Changes to the island landscape also causes adult mortality as turtles are overturned from falling off cliffs, and die from heat exhaustion when trapped by rocky cliffs. By addressing these particular threats through reprofiling of beaches and fencing cliffs, the project has reduced juvenile and adult mortality substantially (Read *et al.* 2018). A similarly targeted approach was used to restore spawning habitat of the spotted handfish (*Brachionichthys hirsutus*) in Tasmania that had been compromised by an invasive species (see Section 7). The critically endangered handfish deposits eggs on semi-rigid structures like kelp, seagrass or the stalked ascidian (*Sycozoa* spp.). These critical spawning habitats have been threatened by swinging yacht moorings, dredging and through predation by the invasive northern pacific seastar (*Asterias amurensis*) (Bruce and Last, 1996; Commonwealth of Australia 2015; Wong 2015; Lynch *et al.* 2016). Conservation efforts include installing new moorings (i.e. threat removal), and deploying ceramic artificial spawning habitats (i.e. artificial habitat creation) with the hopes that this will allow populations to recover.

The introduction of novel threats and the increasing severity of the global biodiversity crisis may increase managers' appetite for novel ideas, including engineered solutions and concepts. It will become increasingly important to be open and receptive to these ideas, and evaluate the costs and potential benefits from restoration. While there are risks and potential unintended consequences associated with any intervention, this should be evaluated against the risk of ongoing decline within the context of current management practices.

## 8.4 Restoration in the context of a changing climate

Climate change is now widely recognised as the preeminent threat against global ecosystems, and this is particularly true for shallow coastal environments. A rapidly changing climate will influence ecological restoration in two ways (1) by stressing and harming previously pristine or undamaged ecosystems, and (2) by permanently changing environmental conditions of degraded ecosystems, such that pristine systems are no longer realistic or appropriate restoration targets.

Climate change has profound consequences for coastal habitats, and the ecosystem services they provide (Harley *et al.* 2006). In particular, because the four habitats covered in this report are primary habitat builders and providers of structural complexity in otherwise relatively barren systems, their loss has cascading implications to other marine organisms. For example, in 2011, a marine heatwave caused the loss of kelp forests from 2300 km of West Australian coastline (Wernberg *et al.* 2016). The enormous habitat loss occurred as a phase shift into a turf algae dominated system, and with an increase in subtropical herbivores that are now suppressing kelp recovery. Despite the removal of the stressor (acute temperature anomaly), the system is showing little sign of recovery. Similarly, seagrasses are vulnerable to changes in temperature, illustrated by a catastrophic die-off of seagrasses in Shark Bay, Western Australia during the 2011 marine heatwave (Fraser *et al*. 2014; Thomson *et al.* 2015). Here, green turtles that rely on the seagrass as a food resource suffered marked health declines following the heatwave. While saltmarshes are less directly affected by increasing temperatures, they are likely to be severely impacted by the increased severity of storms and sea-level rise, concomitant with climate change (Hughes 2011). The effects of climate change on shellfish reefs are relatively unknown, given that many of these habitats have been functionally extinct for decades. Overall, it is clear that many Australian coastal ecosystems are vulnerable to the effects of climate change and that their losses are likely to have cascading effects on the species they support.

Environmental changes brought about by climate change (i.e. increased sea surface temperatures, ocean acidification, sea level rise etc.) are likely to drastically change conditions within coastal habitats. While certain habitats may exhibit some capacity to tolerate changes in environmental conditions, climate change is likely to permanently alter what species can survive in some locations. Ultimately, this may mean that managers have to (1) consider new sites where environmental conditions will be appropriate, (2) shift the targets of restoration away from preferred historical states towards functionally equivalent but taxonomically different systems (Harris *et al.* 2006). An increasing focus on the ecosystem benefits and function may lead to more realistic goals and objectives in this context.

A novel development in the marine restoration space is that of assisted evolution, where remnant survivors from heat events are selectively bred with the hopes that their genetic material may carry a degree of climate resilience (Anthony *et al.* 2017). Alternatively, assisted migration involves the movement of species from one location to another, under the assumption that the novel species will be better suited to cope with the changing environmental conditions. While these techniques are frequently debated in the terrestrial space (McLachlan *et al.* 2007), these techniques are in relatively early stages of investigation for us in the marine habitats.

Finally, removal of the stressor is one of the critical prerequisites of best practice ecological restoration. Therefore, it is critical that restoration interventions are couched within a broader toolkit including meaningful action on climate change. Restoration by itself is not a panacea, as restored ecosystems are likely to still be vulnerable to the broader implications of climate change. Further, the effects of climate change will continue in future decades, even if carbon emissions are reduced to meet the goals of international agreements. However, restoration may be able to buy us time until the climate stabilises by enhancing populations and preserving genetic diversity in the meantime.

## 8.5 Who should pay for restoration?

### 8.5.1 The restoration economy - overseas

The restoration economy describes the economic outputs from ecological restoration, in terms of job creation and industry involvement (BenDor *et al.* 2015), as well as the economic inputs (i.e. who pays for restoration). The US system is perhaps the most similar to the Australian, as they have federal and state management of natural resources. They also have a mature history of restoration projects both in the terrestrial and aquatic realms. There, funding and managing of projects is often a complex relationship of interagency collaboration and public-private partnerships (BenDor *et al.* 2015). It has been estimated that the American federal government invests $1.9 billion per year in restoration-related programs, which is matched and exceed by investments linked to compensatory mitigation (i.e. offsets, $3.8 billion, and non-profit investments $4.3 billion (BenDor *et al*. 2015).

Recently, evidence has been growing that the restoration industry contributes to national economic growth and employment (Figure 8.2). For example, BenDor *et al*. (2015) estimated that up to 33 jobs could be created per US$1 million invested in ecological restoration. In some cases, the benefits of job creation have been estimated to exceed the amount invested in restoration. For example, Kellon and Hesselgrave (2014) calculated that $411.4 million invested in watershed restoration activities in Oregon roughly doubled in terms of economic output and job creation ($752.4-$977.5 million). These estimates do not include valuation of ecosystem services generated, nor the potential costs of environmental damage (from erosion, for example) from a failure to act. These studies highlight how investment in restoration has substantial socio-economic benefits outside of the purely ecological benefits.

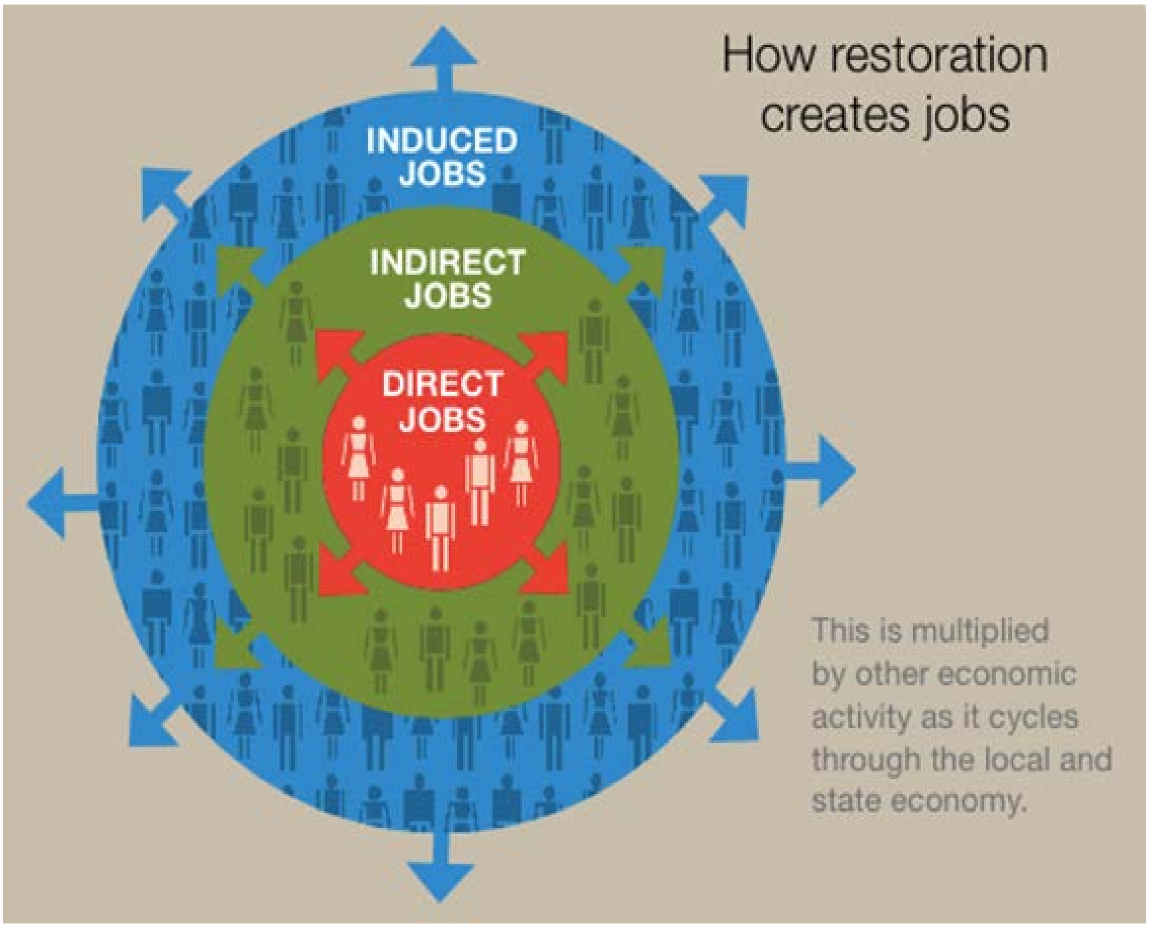


Figure 8.2 Project managers, designers, engineers, construction contractors, technicians, and tourism and education workers – all of these roles represent new jobs, created for the purposes of habitat restoration. Figure courtesy of Restore America's Estuaries.

### 8.5.2 The restoration economy - Australia

Restoration can be expensive including works, stakeholder engagement, monitoring and reporting, and sometimes even land acquisition. It is unlikely that the federal government will have the will or resources to pay for all restoration projects. Therefore state and local government and the private and non-government sectors will also play a key role in financing restoration. Experience from the US has shown that government funding and supportive policy can leverage substantial private funding for large-scale marine and coastal restoration and there are indications that this may also be the case in Australia (Gillies *et al.* 2015).

Australian restoration projects have been funded through a variety of mechanisms. For example, the Raine Island Recovery Project has a budget of almost $8 million funded by a resource company (BHP), the Queensland Government, the Federal Government through the Great Barrier Reef Marine Park Authority and an NGO, the Great Barrier Reef Foundation.

### *Offsets*

Environmental offsets are measures that are designed to compensate for the environmental impacts of an action, after avoidance and mitigation measures are taken. Biodiversity offset policies in the marine environment exist in six countries (Australia, Canada, Columbia, France, Germany, USA; Niner *et al.* 2017). The EPBC Act allows for the provision of offsets to residual impacts to MNES that cannot be avoided or mitigated. Offsets are considered during environmental impact assessments under the EPBC Act and may be required as a condition of approval for a development.

The EPBC Environmental Offsets Policy was produced in 2012 (Australian Government 2012). Although the policy applies to offsetting requirements in terrestrial and aquatic (including marine) environments, they have rarely been enacted under the EPBC Act. For example for the Great Barrier Reef World Heritage Area, marine offsets are managed through the Reef Trust, under outcome 4: *Any new development maintains or improves the condition of matters of national and state environmental significance through the strategic delivery of offsets through the Reef Trust*. An example where offsets have been used under this outcome was a $300,000 investment into the Cairns and Fitzroy Island Turtle Rehabilitation Centre (Grant ID: A0000010951G, MERIT). Payment into the Reef Trust is voluntary, and is based on the Reef Trust Offsets Plan and Calculator. An example of a marine development project, which has made payments into the Reef Trust, is the Abbot Point Growth Gateway project. This project was also required to offset 150 percent of fine sediments released during dredging, through a reduction in the load of fine sediments entering the marine environment from the Burdekin and Don catchments. A challenge with using offsets in the marine realm is that enforcement and compliance may be more difficult due to access to and visibility of habitats.

### *Supporting community-led projects*

Australia has a strong history of supporting community-led restoration and conservation projects through programs such as Landcare and Coastcare and through the Natural Resource Management Agencies. Similar initiatives could provide an opportunity to coordinate community groups into regional networks capable of enacting marine and coastal restoration projects and increasing engagement.

### *Public-private partnerships*

Public-private partnerships are useful for the development of infrastructure in Australia as they allow governments and the private sector to work together and share resources on key projects. A similar approach could be taken for the restoration of natural infrastructure such as marine and coastal habitats. A recent example was the federal government investing

$990,000 through the National Stronger Regions Fund to build native flat oyster reefs as natural infrastructure for local benefits. This project is part the South Australian Government's Blue Infrastructure Initiative which seeks to return these highly productive habitats to coastal waters across the state. The federal investment provided strong leverage for this $4.2 million project with co-investment and support from The Nature Conservancy, South Australian Government, Yorke Peninsula Council and the University of Adelaide.

There are direct links between restoration and industry. For example, costs associated with developing shellfish hatcheries or research and development in shellfish genetics, disease and husbandry could be paid for in part by restoration projects, with industry cost savings returned back to shellfish reef restoration projects.

### *Emission and nutrient trading markets*

As appreciation for, and valuing of, natural systems increases, this opens up opportunities for new ways to pay for conservation or restoration. For instance, the denitrification and phosphorus removal benefits derived from shellfish reefs could provide a nutrient sink mechanism with funding for restoration activities derived from estuarine nutrient trading schemes, sewerage or pollution offsets. Such programs could operate in a similar way to freshwater protection funds, which divert funding from downstream management interventions (e.g. desalination plants) to fund upper catchment restoration projects in order to secure clean water. The fisheries production benefits of shellfish reefs (zu Ermgassen *et al.* 2016) could provide a model for ecosystem-based fisheries management, whereby restoration activities are funded through recreational fisheries license funds or commercial seafood levies.

Coastal habitats like saltmarshes, kelp forests and seagrass meadows play a significant role in sequestering and storing carbon. As carbon markets establish this could be a new way to finance restoration of these systems. An example of this from terrestrial systems is The Reducing Emissions from Deforestation and Forest Degradation (REDD+) mechanism which finances forest restoration activities that contribute to climate change mitigation, sustainable management, and carbon stock enhancement in developing nations (Alexander *et al.* 2011).

### *Insurance*

Recently, innovative approaches to finance protection and restoration of critical ecosystems include insurance against potential damages, so that insurance payouts rapidly funds restoration following a disturbance to the insured ecosystem. For example, the state government of Quintana Roo in Mexico has insured the Mesoamerican Reef, and beach sands of high-density tourism areas against hurricane damage. This was spearheaded by The Nature Conservancy, and Swiss Re, the second largest reinsurance company in the world. These types of insurance solutions are likely to be mainly targeted at critical ecosystem services provided by coastal ecosystems, like shoreline protection, however they may be a novel way to finance costly restoration projects.

## 8.6 Working in partnership with Traditional Owners

A move to more interventions may provide an opportunity to revitalise the partnership between managers and Traditional Owners, as an actively managed system may resonate with traditional Sea Country management. Aboriginal and Torres Strait Islander peoples are important landowners and managers of coastal land and sea Country in Australia through native title bodies, cultural and natural resource management organisations and other corporations (McLeod *et al.* 2018b). Many of the marine and coastal habitats discussed in this report provide traditional food and other resources for Traditional Owners and have been actively managed for 1000s of years. The traditional ecological knowledge gained through this is likely to be critical for restoration success. In 2016, a workshop brought together 21 Traditional Owners from around Australia and New Zealand who were involved in or interested in shellfish restoration projects to share their advice about best practices to work in partnership in shellfish restoration projects. Figure 8.3 shows their seven most important pieces of advice and are likely to be general to any marine and coastal restoration projects.


Figure 8.3: The seven pearls of wisdom produced by Traditional Owners. Figure used with permission from McLeod *et al.* 2018b.

## 8.7 The value of national coordination

Ecological restoration has some recurring problems and challenges that limit success. These have been outlined in many reviews and reports (e.g. Lake 2001; Suding 2011) and include:

- A general lack of relevant long-term monitoring (or no monitoring at all)
- Lack of clear objectives
- Poor communication of objectives
- Projects working in isolation, repeating past mistakes
- Poor site selection
- Disconnect between the values and needs of scientists and practitioners
- Stressor not removed

It is possible that marine and coastal restoration as a relatively new initiative in Australia could be affected by the same problems. Leadership and co-ordination from the Commonwealth could encourage better practices. The following are suggestions for what shape and content such leadership could provide.

1. Current issue: *It is challenging to know what part of the DoEE to contact.* In particular, restoration proponents would likely benefit from collaboration between Federal and State departments.
    - Potential solution: Develop a point of contact team (possibly as part of the partnerships team?).

2. Current issue: *Lack of adequate monitoring*.
    - Potential solution: Commonwealth Government could support states by producing best practice monitoring protocols or requiring monitoring through permit processes.

3. Current issue: *Lack of clear objectives, poor methods and site selection.*
    - Potential solution: National coordination group should produce and provide best practice guidelines on how to plan, develop and implement restoration projects and reward best practice projects with co-investment.

4. Current issue: *projects working in isolation*
    - Potential solution: Restoration coordination group acts as the hub for knowledge sharing between Australian restoration projects, and with international restoration groups.

Coastal areas are important to a wide-range of stakeholders, who often have competing interests and this requires effective management. Local, state and national government have roles in decisions relating to coastal management. One of the benefits of marine and coastal restoration is that it brings together a wide-range of stakeholders, some of which have been in conflict in the past, in a positive way. For example, the Shellfish Reef Restoration Network includes conservation groups and recreational fishers, who are often in an adversarial relationship over issues such as marine park creation.

## 8.8 Recommendations

*Consider all options for the recovery of MNES*

- *Consider all potential management actions for the recovery of MNES*. Consider threat reduction, habitat protection (reserves) and active interventions, and weigh up potential costs and benefits of these approaches when considering management actions to preserve MNES. Restoration of marine and coastal habitats once seemed too difficult to be a useful management too, but this report has demonstrated that restoration is likely to be a useful tool in many instances.
- *Consider restoration as a complementary tool to other management actions*. Restoration does not necessarily need to be undertaken sequentially or separately from other management activates, or only as an offset requirement. It can occur in parallel as it can complement and provide positive feedback loops. Good examples where interventions are part of a suite of management actions include the Raine Island Recovery Project and the spotted handfish project in Tasmania. On Raine Island, turtle nesting habitat restoration has been couched within broader management actions like pollution control, habitat protection and fisheries management (Commonwealth of Australia 2017). In Tasmania, artificial spawning habitats have been deployed to protect the spotted handfish, alongside threat reduction activities like the installation of new moorings to reduce habitat damage from anchors (See Case Study, Section 7 for further details).

*Consider the risk of inaction, as well as action*

- *The risk of action should be weighed against the risk of inaction.* Currently, conservation is often viewed through a lens of preservation or protection of pristine systems, where any habitat-modifying intervention to that system is considered a risk. Thus, a large-scale restoration project is weighed on the same scale as substantial development projects. However, given that all four ecosystems in this report have experienced substantial declines under current management strategies, it is clear that novel solutions should be considered. Risk should clearly be taken into account when considering restoration actions; however, the risk of not taking restorative actions should also be considered in these assessments.

*Develop a policy pathway to restoration*

- *Modify the interpretation of existing policy or develop fit-for-purpose policy to distinguish restoration from development*. As a relatively new initiative in Australia, ecological restoration does not have a history of targeted policy in the marine and coastal environment. Any proposed restoration activity is therefore judged based on policy that may be a poor fit for the activity For example, providing of natural reefs for shellfish or kelp habitats is currently regulated under the *Environmental Protection (Sea Dumping) Act* 1981 (C'th)). In some circumstances only a shift in definition or permit process is needed for this change, however, new policy may need to be developed in the medium to long-term.
- *Streamline permitting for marine and coastal habitat restoration projects.* In contrast to terrestrial restoration projects, marine and coastal restoration projects almost always occur

on Crown land, therefore, governments (at a federal, state or local level) are required to be more involved in and have more of an interest in the proposed restoration activities. In addition, permit proponents need to consider other regulatory, insurance and safety issues such as working in or near water (e.g. diving and marine biosecurity protocols). These factors along with permitting processes that are not fit-for-purpose mean that timelines from conception to implementation are therefore generally much slower compared to terrestrial projects, which places a heavier financial burden on proponent. This ends up being cost prohibitive for smaller groups.

- *Enable permitting processes to weight overall benefits, costs and risk*. Permitting processes and culture could be refined to weigh the overall potential benefits of a project with risk of small-scale damage. For example, it is very difficult to get a permit for shellfish reef restoration if there is seagrass present, even if the seagrass is in poor condition and located in an area with historical evidence of shellfish reefs. Further, saltmarsh restoration projects that require the removal of encroaching mangroves can be blocked due to their protected status in some states.
- *Use permit process to ensure best practice procedures*. Encourage appropriate planning, and monitoring to encourage best practice ecological restoration. This could be achieved by assessing the appropriateness of the restoration actions for the stated objectives and requiring appropriate monitoring and reporting on the progress and outcomes of projects.
- *Enable new funding opportunities for restoration.* Develop and support funding pathways to encourage restoration projects through offsets, environmental insurance, private-public partnerships, and community led volunteer projects, and co-investment states and local government. See section 8.15 for more details on funding opportunities.

### *Value ecosystem services of blue infrastructure*

- *Prioritise research to estimate the economic value of habitats.* Marine and coastal habitats can provide numerous benefits such as supporting fish productivity, carbon sequestration, nutrient cycling, coastal protection and recreation. Decision makers need to be able to weigh up the relative costs and benefits of coastal development, protection or restoration and to do this they need robust, accessible and defensible data on the ecological function and economic value of habitats and the ecosystem services they provide. If no quantified value is available, the risk is that the value of ecosystem services are unlikely to be included in decisions. The Federal Government has already put considerable investment into this through NESP and the Fisheries Research and Development Corporation and other initiatives and future investment could build on this base. Ongoing research could feed into the National Strategy for Environmental-Economic Accounting (Australian Government 2018).

### *Consider recent history and plan for a changing climate*

- *Historical assessments should be included when setting baselines for protection-focused management.* For example, the Corner Inlet Ramsar wetland contained extensive shellfish reefs, but these were not mentioned in the initial site assessment and therefore are unlikely to be included in management plans. Permitting processes may then defend the status quo rather than other historical or desired states.

- *Challenge the assumption that protected areas are pristine.* It is a natural presumption that protected areas are in a pristine condition, and therefore not appropriate sites for restoration. However, protected areas where many stressors are removed may be ideal areas for restoration, and restoration may be needed to preserve the value of protected areas.
- *Challenge the assumption that restoration will restore areas to being pristine*. In most cases it will not be possible (nor desired, considering the changing climate) to return to a pre-impact ‘pristine’ or historical state. Many restoration projects focus now on restoring critical ecosystem function and services. For example a restored kelp forest may not be identical to the historical state, but is likely to support a more productive and biodiversity system than the urchin barren it replaced.

### *Invest in knowledge sharing, collaboration and best practice guidance*

- *Learn from international experience*. Work with other countries that have longer histories with restoration to learn from their experiences. Lessons from overseas could be used to inform policy, decision-making tools, workflows and best practice guidelines in Australia.
- *Build on the Blue Carbon Initiative to include other habitats*. The Blue Carbon Initiative is a good example of an international initiative where Australia is providing leadership. This could be expanded to include other marine and coastal habitats such as kelp forest so that they can be included in plans to protect and restore marine and coastal ecosystems for their ‘Blue Carbon’ value.
- *Build capacity in partner nations for restoration of marine and coastal habitats.* Especially when these are linked to food security, alternative livelihoods, and shoreline protection. This could be an important component of Australian foreign aid in the future that may be more cost-effective than investing in built infrastructure.
- Support marine and coastal habitat restoration network. Networks such as the Shellfish Reef Restoration Network, the Seagrass Restoration Network and the Coastal Restoration Network may provide useful contacts to assist with development of national policy, recovery plans and disseminating best practices. Notable here is the absence of kelp forests, as such there appears to be considerable need to establish a national network to coordinate and facilitate the restoration of Australian kelp habitats.
- *Consider positive feedback loops, and system-wide restoration approaches*. Projects could target a variety of habitats, and the stressors causing their decline within a system rather than just addressing each habitat and threat separately. For example, water quality improvement through wastewater treatment upgrades could be matched with active habitat restoration. In addition, there can be positive feedback loops between and within habitats. For example, oyster reef restoration can encourage the growth of seagrass meadows nearby (Wall *et al.* 2008), and healthy seagrass meadows are associated with less disease in nearby coral reefs (Lamb *et al.* 2017), and the presence of transplanted adult kelp facilitate the recruitment of juveniles of the same species (Layton *et al.* in review).
- *Investing in larger projects to attract more co-investment.* Government investment is likely to encourage buy-in from a wider range of stakeholders and may attract funding from new sources. For example, the $990,000 investment into shellfish reef restoration in South

Australia as ‘natural infrastructure’ help encourage co-investment for the >$4 million project.

- *Avoid spreading funding and effort too thin*. Small projects are important because they provide the research and development necessary for scale-up, and often include many community stakeholders. However, underfunding many small projects may reduce the likelihood of success, and some types of restoration may only succeed at a larger scale.

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# APPENDIX A – HISTORICAL DECLINES OF SEAGRASS

| Timeframe | Location | Area of loss (ha)* | *Species* | Drivers of loss | Reference |
|---|---|---|---|---|---|
| *New South Wales* | | | | | |
| 1930-1999 | Gunnamata Bay | 16 | *Posidonia australis, Zostera muelleri* | Severe storms; Bait digging; | Williams and Meehan 2001 |
| 1930-1985 | Botany Bay | - | *Posidonia australis* | Catchment management; Grazing (sea urchin) | Larkum and West 1990 |
| 1942-1999 | Burraneer Bay | 5 | *Posidonia australis, Zostera muelleri* | Dredge disposal | Williams and Meehan 2001 |
| 1942-1999 | Cabbage Tree Basin | 12 | *Posidonia australis, Zostera muelleri, Halophila sp.* | Sand migration | Williams and Meehan 2001 |
| 1948-1994 | Merimbula Lake | 47 | *Posidonia australis* | - | Meehan 1997 |
| 1951-1999 | Lilli Pilli Point | 7 | *Posidonia australis, Zostera muelleri* | Channel Dredging and sand migration | Williams and Meehan 2001 |
| 1951-1999 | Red Jacks Point | - | *Posidonia australis* | Shell grit excavation | Williams and Meehan 2001 |
| 1957-1994 | Wagonga Inlet | 8 | *Posidonia australis* | - | Meehan and West 2002 |
| 1957-1998 | Bermagui River | 14 | *Posidonia australis* | - | Meehan and West 2002 |
| 1960-1989 | Jervis Bay | - | *Posidonia australis.* | Seismic testing | West *et al.* 1989 |
| 1961-1998 | St Georges Basin | 86 | *Posidonia australis* | - | Meehan and West 2002 |
| 1981-1997 | Brisbane Waters | 258 | - | - | Williams *et al.* 2003 |
| 1988-1995 | Wallis Lake | 518 | *Posidonia australis* | - | Dekker *et al.* 2005 |

| Timeframe | Location | Area of loss (ha)* | *Species* | Drivers of loss | Reference |
|---|---|---|---|---|---|
| ____-1986 | Lake Macquarie | 700 | *Zostera capricorni**, Halophila ovalis, Ruppia megacarpa* | Light reduction (eutrophication) | King and Hodgson 1986 |
| ____-1986 | Tuggerah Lakes | 1,300 | *Zostera capricorni,**, Halophila ovalis, Ruppia megacarpa* | Light reduction (eutrophication) | King and Hodgson 1986 |
| *Northern Territory* | | | | | |
| 1985-1985 | West Island - Limmen Bight | 18,300 | *Halodule uninervis; Halophila ovalis; Syringodium isoetifolium; Cymodocea serrulata; Halophila spinulosa* | Cyclone | Poiner *et al.* 1987 |
| 1991-1992 | Torres Strait | 10,000 | *Halodule uninervis; Halophila ovalis; Syringodium isoetifolium; Cymodocea serrulata; Halophila spinulosa* | - | CSIRO study reported in Kirkman *et al.* 1999 |
| *Queensland* | | | | | |
| 1992-1993 | Hervey Bay | 100,000 | *Zostera capricorni*** | Flooding; Cyclone | Preen *et al.* 1995 |
| 1995-2012 | Lizard Island | 8 | *Thalassia hemprichii, Halodule uninervis* | Eutrophication | Saunders *et al.* 2015 |
| 1998-2001 | Urangan | 91 | *Zostera capricorni*** | - | Campbell and McKenzie 2004. |

| Timeframe | Location | Area of loss (ha)* | *Species* | Drivers of loss | Reference |
|---|---|---|---|---|---|
| 1998-1999 | Wanggoolba Creek | 120 | *Zostera capricorni*** | - | Campbell and McKenzie 2004. |
| 1998-2002 | Northern Great Sandy Strait | 1,896 | *Zostera capricorni*** | - | Campbell and McKenzie 2004. |
| 2001-2013 | Townsville | 3,700 | *Zostera muelleri, Halophila ovalis, Halophila spinulosa* | Cyclone | Coles *et al.* 2015 |
| 2002-2013 | Gladstone | 1,600 | *Zostera muelleri, Halophila ovalis, Halodule uninervis* | Flooding, dredging, land reclamation | Coles *et al.* 2015 |
| 2007-2011 | Southport Broadwater Parklands | 1 | *Zostera muelleri, Halophila ovalis* | Land reclamation | Hall 2011 |
| 2007-2013 | Cairns | 700 | *Zostera muelleri, Halophila ovalis* | - | Coles *et al.* 2015 |
| 2007-2007 | Broadwater | 1 | *Zostera muelleri, Halophila ovalis* | Land reclamation | BIOME 2007 |
| ____-1990 | Moreton Bay | 257 | *Zostera capricorni*** | Sediment burial | Kirkman 1978 |
| *South Australia* | | | | | |
| 1908-1914 | Port Broughton | 320 | *Posidonia australis, Posidonia sinuosa* | Fibre harvesting | Irving 2013 |
| 1932-1975 | Proper Bay | 38 | *Posidonia australis* | Nutrient enrichment (meat-works discharge) | Shepherd 1986 |
| 1939-present | Whyalla | 2,000 | *Posidonia* | Channel dredging; industrial discharge (ammonia) | Shepherd 1986 Harbison and Wiltshire 1993 |

| Timeframe | Location | Area of loss (ha)* | *Species* | Drivers of loss | Reference |
|---|---|---|---|---|---|
| 1949-1995 | Adelaide | 4,000 | *Posidonia sinuosa, Amphibolis antarctica* | Eutrophication | EPA 1998 |
| 1967-1967 | Mambray creek to Douglas Point | 14 | *Posidonia australis* | Pipeline construction | Irving 2014 |
| 1978-1985 | Gulf of St Vincent | 365 | *Posidonia spp. , Amphibolis spp.* | Sewage outfall Eutrophication | Neverauskas 1987 |
| 1993-1993 | Spencer Gulf | 12,717 | *Amphibolis antarctica and intertidal Zostera spp* | Thermal stress | Seddon *et al.* 2000 |
| ____-1985 | Holdfast Bay and off Bolivar | 5,222 | *Posidonia sinuosa, Posidonia angustifolia, Amphibolis antarctica* | Light reduction Increased epiphytism Eutrophication | Neverauskas 1987 |
| ____-1988 | South of Outer harbour, Holdfast Bay | 100 | *Posidonia sinuosa* | Sediment burial | Sergeev *et al.* 1988 |
| ____-1988 | Holdfast Bay | - | *Posidonia sinuosa, Posidonia angustifolia* | Sediment instability (blow outs) | Sergeev *et al.* 1988 |
| ____-1949 | Gulf St Vincent | 900 | *Heterozostera tasmanica; Posidonia sinuosa; Amphibolis antarctica* | Coastal construction (retaining walls, groynes), sediment burial. | Shepherd *et al.* 1989 |
| *Tasmania* | | | | | |
| 1948-1990 | Birch Point | 397 | | | Hamdorf and Kirkman 1995 |

| Timeframe | Location | Area of loss (ha)* | *Species* | Drivers of loss | Reference |
|---|---|---|---|---|---|
| 1948-1990 | Ralphs Bay | 430 | *Zostera muelleri, Heterozostera tasmanica, Halophila australis* | | Hamdorf and Kirkman 1995 |
| 1948-1990 | Pittwater | 1,201 | *Zostera muelleri, Heterozostera tasmanica, Halophila australis* | | Hamdorf and Kirkman 1995 |
| 1970-1990 | Norfolk Bay | 2,140 | *Zostera muelleri, Heterozostera tasmanica, Halophila australis* | | Hamdorf and Kirkman 1995 |
| | Clarence River | 445 | | | Hamdorf and Kirkman 1995 |
| *Victoria* | | | | | |
| ____-1983 | Western Port | 17,800 | *Heterozostera tasmanica, Zostera muelleri* | Sedimentation of fine muds | Bulthuis 1983 |
| *Western Australia* | | | | | |
| 1941-1992 | Rottnest Island | 46 | *Posidonia sinuosa, Posidonia australis* | Mooring damage-erosion- | Hastings *et al.* 1995 |
| 1953-2002 | Warnbro Sound | 73 | *Posidonia australis* | Sediment movement | Bridgwood 2006 |
| 1956-2001 | Esperance Bay | 132 | *Posidonia australis* | Port and coastal construction | Hegge and Kendrick 2005 |
| -1986 | Cockburn Sound | 2,268 | *Posidonia sinuosa, Posidonia australis* | Light reduction Increased epiphytism | Kendrick *et al.* 2002; Cambridge and McComb 1984 |
| -1986 | Princess Royal Harbour | 810 | Posidonia australis, Amphibolis antarctica | Light reduction Increased epiphytism (eutrophication) | Bastyan 1986 |

| Timeframe | Location | Area of loss (ha)* | *Species* | Drivers of loss | Reference |
|---|---|---|---|---|---|
| -1986 | Oyster Harbour | 720 | Posidonia australis, Amphibolis antarctica | Light reduction Increased epiphytism (eutrophication) | Bastyan 1986 |
| 2011-2014 | Shark Bay | ***86, 000 | Amphibolis antarctica | Thermal stress combined with light stress | Fraser *et al.* 2014; Thomson *et al.* 2015; Ariaz-Ortiz et al 2018 |

# APPENDIX B – AUSTRALIAN SEAGRASS RESTORATION ATTEMPTS

| ID | **Citation** | **Year** | Genus | Species | Location | State | Transplant unit | Environment | Tide | Wave exposure | Experiment | Enhancement | Duration (years) | Success (%) | Objective | Depth |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | **Meehan and West 2002** | **2002** | *Posidonia* | *australis* | Port Hacking, Gunnamatta Bay | NSW | Sprig | estuary | subtidal | Sheltered | 1 | 0 | <1 | 0 | | shallow |
| 2 | **Meehan and West 2002** | **2002** | *Posidonia* | *australis* | Port Hacking, Lilli Pilli Point | NSW | Sprig | estuary | subtidal | Sheltered | 1 | 0 | <1 | 0 | | shallow |
| 3 | **Meehan and West 2002** | **2002** | *Posidonia* | *australis* | Port Hacking, Burraneer Bay | NSW | Sprig | estuary | subtidal | Sheltered | 1 | 0 | >1 | 100 | | shallow |
| 4 | **Bastyan and Cambridge 2008** | **2008** | *Posidonia* | *australis* | Oyster Harbour | WA | Sprig | estuary | subtidal | Sheltered | 0 | 0 | >5 | 0 | Pilot to test feasibility and anchoring | 1m |
| 5 | **Bastyan and Cambridge 2008** | **2008** | *Posidonia* | *australis* | Oyster Harbour | WA | Sprig | estuary | subtidal | Sheltered | 1 | 1 | >5 | 98 | Pilot to test feasibility and anchoring | 1m |
| 6 | **Campbell and Paling 2003** | **2003** | *Posidonia* | *australis* | Success Bank | WA | Plug | Marine | subtidal | High | 1 | 1 | 1 | 42 | trial of habitat enhancement | 9m |
| 7 | **Paling *et al.* 2000** | **2001** | *Posidonia* | *sinuosa* | Cockburn Sound | WA | Plug | Marine | subtidal | Medium | 1 | 0 | 2 | 77 | Research | |
| 8 | **Paling *et al.* 2000** | **2001** | *Posidonia* | *australis* | Cockburn Sound | WA | Plug | Marine | subtidal | Medium | 1 | 0 | 2 | 76 | Research | |
| 9 | **Paling *et al.* 2000** | **2001** | *Amphibolis* | *griffthii* | Cockburn Sound | WA | Plug | Marine | subtidal | Medium | 1 | 0 | 2 | 44 | Research | |

| ID | **Citation** | **Year** | Genus | Species | Location | State | Transplant unit | Environment | Tide | Wave exposure | Experiment | Enhancement | Duration (years) | Success (%) | Objective | Depth |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 10 | **Wear *et al.* 2010** | **2010** | *Amphibolis* | *griffthii* | Holdfast Bay | SA | Seeds | Marine | subtidal | High/Medium | 1 | 1 | 1 | 31 | | 1m |
| 11 | **Wear *et al.* 2010** | **2010** | *Amphibolis* | *antarctica* | Holdfast Bay | SA | Seeds | Marine | inter and sub tidal | High/Medium | 1 | 1 | 1 | 31 | | 1m |
| 12 | **Wear *et al.* 2010** | **2010** | *Amphibolis* | *antarctica* | Holdfast Bay | SA | Seeds | Marine | inter and sub tidal | High/Medium | 1 | 1 | 3 | 7 | | 1m |
| 13 | **Wear *et al.* 2010** | **2010** | *Amphibolis* | *antarctica* | Holdfast Bay | SA | Seeds | Marine | inter and sub tidal | High/Medium | 1 | 1 | 3 | 7 | | 1m |
| 14 | **LEC 1990 in Lord *et al.* 1999*et al.*** | **1990** | *Posidonia* | *australis* | Success and Parmelia Banks | WA | Seedlings | Marine | subtidal | High/Medium | 1 | 0 | <1 | 0 | Research | 7m-20m |
| 15 | **LEC 1990 in Lord *et al.* 1999** | **1990** | *Posidonia* | *australis* | Owen Anchorage | WA | Seedlings | Marine | subtidal | High/Medium | 1 | 0 | <1 | 0 | Research | 7m-20m |
| 16 | **Cambridge 1980 in Lord *et al.* 1999** | **1980** | *Posidonia* | *australis* | Cockburn sound | WA | Plug | Marine | subtidal | Medium | 1 | 0 | <1 | 35 | Research | 2-3m |
| 17 | **Cambridge 1980 in Lord *et al.* 1999** | **1980** | *Posidonia* | *sinuosa* | Warnbro Sound | WA | Plug | Marine | subtidal | Medium | 1 | 0 | <1 | 35 | Research | 2-3m |
| 18 | **Hancock 1992 in Lord *et al.* 1999** | **1992** | *Posidonia* | *australis* | Carnac Island | WA | Seedlings | Marine | subtidal | High/Medium | 1 | 0 | 1 | 12 | Research | 3m-11m |
| 19 | **Hancock 1992 in Lord *et al.* 1999** | **1992** | *Posidonia* | *sinuosa* | Carnac Island | WA | Seedlings | Marine | subtidal | High/Medium | 1 | 0 | 1 | 12 | Research | 3m-11m |
| 20 | **Hancock 1992 in** | **1992** | *Posidonia* | *coriacea* | Carnac Island | WA | Seedlings | Marine | subtidal | High/Medium | 1 | 0 | 1 | 12 | Research | 3m-11m |

| ID | **Citation** | **Year** | Genus | Species | Location | State | Transplant unit | Environment | Tide | Wave exposure | Experiment | Enhancement | Duration (years) | Success (%) | Objective | Depth |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Lord *et al.* 1999** | | | | | | | | | | | | | | | |
| 21 | **Hancock 1992 in Lord *et al.* 1999** | **1992** | *Amphibolis* | *griffthii* | Carnac Island | WA | Seedl ings | Marine | subtidal | High/Medium | 1 | 0 | 1 | 12 | Research | 3m-11m |
| 22 | **Hancock 1992 in Lord *et al.* 1999** | **1992** | *Posidonia* | *australis* | Carnac Island | WA | Sprig | Marine | subtidal | High/Medium | 1 | 0 | 1 | 44 | Research | 3m-11m |
| 23 | **Hancock 1992 in Lord *et al.* 1999** | **1992** | *Posidonia* | *sinuosa* | Carnac Island | WA | Sprig | Marine | subtidal | High/Medium | 1 | 0 | 1 | 44 | Research | 3m-11m |
| 24 | **Hancock 1992 in Lord *et al.* 1999** | **1992** | *Posidonia* | *coriacea* | Carnac Island | WA | Sprig | Marine | subtidal | High/Medium | 1 | 0 | 1 | 44 | Research | 3m-11m |
| 25 | **Hancock 1992 in Lord *et al.* 1999** | **1992** | *Amphibolis* | *griffthii* | Carnac Island | WA | Sprig | Marine | subtidal | High/Medium | 1 | 0 | 1 | 44 | Research | 3m-11m |
| 26 | **Paling 1992 in Lord *et al.* 1999** | **1992** | *Posidonia* | *australis* | Shoalwater Bay | WA | Sprig | Marine | subtidal | Sheltered | 1 | 0 | 3 | 0 | Research | 5m |
| 27 | **Paling 1992 in Lord *et al.* 1999** | **1992** | *Posidonia* | *sinuosa* | Shoalwater Bay | WA | Sprig | Marine | subtidal | Sheltered | 1 | 0 | 3 | 0 | Research | 5m |
| 28 | **Walker 1994 in** | **1994** | *Posidonia* | *sinuosa* | Carnac Island | WA | Plug, Sprig | Marine | subtidal | High | 1 | 1 | >1 | 60 | Research | 6m |

| ID | **Citation** | **Year** | Genus | Species | Location | State | Transplant unit | Environment | Tide | Wave exposure | Experiment | Enhancement | Duration (years) | Success (%) | Objective | Depth |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Lord *et al.* 1999** | | | | | | | | | | | | | | | |
| 29 | **Walker 1994 in Lord *et al.* 1999** | **1994** | *Amphibolis* | *griffthii* | Carnac Island | WA | Plug, Sprig | Marine | subtidal | High | 1 | 1 | >1 | 60 | Research | 6m |
| 30 | **Nelson unpublished in Lord *et al.* 1999** | | *Posidonia* | *sinuosa* | Success Bank | WA | Plug | Marine | subtidal | High | 1 | 1 | 3 | 60 | Research | 5-12m |
| 31 | **Nelson unpublished in Lord *et al.* 1999** | | *Posidonia* | *coriacea* | Success Bank | WA | Plug | Marine | subtidal | High | 1 | 1 | 3 | 50 | Research | 5-12m |
| 32 | **Walker 1985 in Lord *et al.* 1999** | | *Amphibolis* | *antarctica* | Rottness Island | WA | Seedlings | Marine | inter and sub tidal | High/Medium | 1 | 0 | >1 | 15 | Research | <6m |
| 33 | **Walker 1985 in Lord *et al.* 1999** | | *Amphibolis* | *griffthii* | Rottness Island | WA | Seedlings | Marine | subtidal | High/Medium | 1 | 0 | >1 | 15 | Research | <6m |
| 34 | **Paling unpublished in Lord *et al.* 1999** | | *Amphibolis* | *griffthii* | Success Bank | WA | Plug | Marine | subtidal | High | 1 | 0 | >1 | 40 | Research | 5-15m |
| 35 | **Walker unpublished in Lord *et al.* 1999** | | *Amphibolis* | *antarctica* | Cockburn Sound | WA | Plug Seedlings | Marine | inter and sub tidal | Medium | 1 | 0 | >1 | 0 | Research | <6m |
| 36 | **Walker unpublish** | | *Amphibolis* | *antarctica* | Warnbro Sound | WA | Plug Seedlings | Marine | inter and sub tidal | Medium | 1 | 0 | >1 | 100 | Research | <6m |

| ID | **Citation** | **Year** | Genus | Species | Location | State | Transplant unit | Environment | Tide | Wave exposure | Experiment | Enhancement | Duration (years) | Success (%) | Objective | Depth |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **ed in Lord *et al.* 1999** | | | | | | | | | | | | | | | |
| 37 | **Paling unpublished in Lord *et al.* 1999** | | *Amphibolis* | *griffthii* | Success bank | WA | Plug | Marine | subtidal | High | 1 | 0 | >1 | 40 | Research | 5-15m |
| 38 | **Paling unpublished in Lord *et al.* 1999** | | *posidonia* | *coriacea* | Success Bank | WA | Sods | Marine | subtidal | High | 1 | 0 | >1 | 95 | Research | 5m |
| 39 | **Paling unpublished in Lord *et al.* 1999** | | *Amphibolis* | *griffthii* | Success Bank | WA | Sods | Marine | subtidal | High | 1 | 0 | >1 | 95 | Research | 5m |
| 40 | **Paling unpublished in Lord *et al.* 1999** | | *Posidonia* | *sinuosa* | Success Bank | WA | Sods | Marine | subtidal | High | 1 | 0 | >1 | 95 | Research | 5m |
| 41 | **Bastyan and Cambridge 2008** | **2008** | *Posidonia* | *sinuosa* | Princess Royal Harbour | WA | Sprig | Marine | subtidal | High | 1 | 1 | >5 | 95 | Research | 5m |
| 42 | **Kirkman 1998** | **1998** | *Posidonia* | *australis* | Cockburn sound | WA | Sprig | Marine | subtidal | High/Medium | 1 | 0 | >2 | 4 | Research | 5m |
| 43 | **Kirkman 1998** | **1998** | *Posidonia* | *sinuosa* | Cockburn sound | WA | Sprig | Marine | subtidal | High/Medium | 1 | 0 | >2 | 4 | Research | 5m |
| 44 | **Kirkman 1998** | **1998** | *Posidonia* | *coriacea* | Cockburn sound | WA | Sprig | Marine | subtidal | High/Medium | 1 | 0 | >2 | 4 | Research | 5m |
| 45 | **Kirkman 1995 in Lord *et al.* 1999** | **1995** | *Amphibolis* | *antarctica* | Cockburn sound | WA | Mats | Marine | inter and sub tidal | High/Medium | 1 | 0 | >2 | 15 | Research | 5m |

| ID | **Citation** | **Year** | Genus | Species | Location | State | Transplant unit | Environment | Tide | Wave exposure | Experiment | Enhancement | Duration (years) | Success (%) | Objective | Depth |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 46 | **Kirkman 1995 in Lord *et al.* 1999** | **1995** | *Amphibolis* | *antarctica* | Cockburn sound | WA | Sprig | Marine | inter and sub tidal | High/Medium | 1 | 0 | >2 | 0 | Research | 5m |
| 47 | **Kirkman 1995 in Lord *et al.* 1999** | **1995** | *Amphibolis* | *griffthii* | Cockburn sound | WA | Mats | Marine | subtidal | High/Medium | 1 | 0 | >2 | 15 | Research | 5m |
| 48 | **Kirkman 1995 in Lord *et al.* 1999** | **1995** | *Amphibolis* | *antarctica* | Warnbro Sound | WA | Mats | Marine | inter and sub tidal | High/Medium | 1 | 0 | <1 | 24 | Research | 5m |
| 49 | **Kirkman 1995 in Lord *et al.* 1999** | **1995** | *Amphibolis* | *griffthii* | Warnbro Sound | WA | Mats | Marine | subtidal | High/Medium | 1 | 0 | <1 | 24 | Research | 5m |
| 50 | **Kirkman 1998** | **1998** | *Posidonia* | *coriacea* | Rottnest Island | WA | Sprig | Marine | subtidal | High/Medium | 1 | 0 | >2 | 35 | Research | 5m |
| 51 | **Kirkman 1998** | **1998** | *Posidonia* | *australis* | Rottnest Island | WA | Sprig | Marine | subtidal | High/Medium | 1 | 0 | >2 | 35 | Research | 5m |
| 52 | **Kirkman 1998** | **1998** | *Posidonia* | *augustifol ia* | Rottnest Island | WA | Sprig | Marine | subtidal | High/Medium | 1 | 0 | >2 | 3 | Research | 5m |
| 53 | **Kirkman 1995 in Lord *et al.* 1999** | **1995** | *Amphibolis* | *antarctica* | Rottnest Island | WA | Mats | Marine | inter and sub tidal | High/Medium | 1 | 0 | >2 | 3 | Research | 5m |
| 54 | **Kirkman 1995 in Lord *et al.* 1999** | **1995** | *Amphibolis* | *griffthii* | Rottnest Island | WA | Mats | Marine | subtidal | High/Medium | 1 | 0 | >2 | 3 | Research | 5m |
| 55 | **West *et al.* 1990; West 1995** | **1990; 1995** | *Posidonia* | *australis* | Lady Robinson Beach | NSW | Sprig | Marine | subtidal | Site subject to increasing | 1 | 1 | <1 | 0 | Rehabilitation | 1m2m and 3m |

| ID | **Citation** | **Year** | Genus | Species | Location | State | Transplant unit | Environment | Tide | Wave exposure | Experiment | Enhancement | Duration (years) | Success (%) | Objective | Depth |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **in Lord *et al.* 1999** | | | | | | | | | wave action from dredging | | | | | | |
| 56 | **West *et al.* 1990; West 1995 in Lord *et al.* 1999** | **1990; 1996** | *Posidonia* | *australis* | Botany Bay | NSW | Sprig | Marine | subtidal | Site subject to increasing wave action from dredging | 1 | 1 | <1 | 0 | Rehabilitation | 1m2m and 3m |
| 57 | **West *et al.* 1990; West 1995 in Lord *et al.* 1999** | **1990; 1997** | *Zostera* | *capricorni* | Lady Robinson Beach | NSW | Sprig | Marine | inter and sub tidal | Site subject to increasing wave action from dredging | 1 | 1 | <1 | 0 | Rehabilitation | 1m2m and 3m |
| 58 | **West *et al.* 1990; West 1995 in Lord *et al.* 1999** | **1990; 1998** | *Zostera* | *capricorni* | Botany Bay | NSW | Sprig | Marine | inter and sub tidal | Site subject to increasing wave action from dredging | 1 | 1 | <1 | 0 | Rehabilitation | 1m2m and 3m |
| 59 | **Gibbs 1997 in Lord *et al.* 1999** | **1997** | *Zostera* | *capricorni* | Botany Bay | NSW | n/a | Marine | inter and sub tidal | Sheltered | 0 | 1 | | 0 | Recolonisation | |
| 60 | **West unpublish ed in Lord *et al.* 1999** | **1999** | | | Hastings River | NSW | Sods | estuary | inter and sub tidal | Sheltered | 1 | 0 | 1 | 100 | Creation | 1-2m below ISLW |
| 61 | **West unpublish ed in Lord *et al.* 1999** | **1999** | | | Hastings River | NSW | Sods | estuary | inter and sub tidal | Sheltered | 1 | 0 | 1 | 0 | Creation | 1-2m below ISLW |
| 62 | **QDPI unpublish ed; Long unpublish** | | *Zostera* | *capricorni* | Cairns Harbour | QLD | Plug | estuary | inter and sub tidal | Sheltered | 1 | 0 | <1 | 100 | Research | 0.5m above MSL |

| ID | **Citation** | **Year** | Genus | Species | Location | State | Transplant unit | Environment | Tide | Wave exposure | Experiment | Enhancement | Duration (years) | Success (%) | Objective | Depth |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **ed in Lord *et al.* 1999** | | | | | | | | | | | | | | | |
| 63 | **Poiner and Conacher 1992; Conacher et al1993; Kenyon pers. comm. in Lord *et al.* 1999** | **1992** | *Zostera* | *capricorni* | Raby Bay | QLD | Turfs | estuary | intertidal | Sheltered | 1 | 1 | >1 | | Research | 2m |
| 64 | **Poiner and Conacher 1992; Conacher et al 1993; Kenyon pers. comm. in Lord *et al.* 1999** | **1992** | *Zostera* | *capricorni* | Victoria Point | QLD | Turfs | estuary | intertidal | Sheltered | 1 | 1 | >1 | | Research | 2m |
| 65 | **Poiner *et al.* 1987; Thorogood *et al.* 1990; Kenyon pers. comm. in** | **1987; 1990** | *10 species including species of Cymodocea Syringodium Halodule and Halophila* | | Western Gulf of Carpentaria | NT | Sprig | Marine | subtidal | | 0 | 0 | 1 | 20 | Monitoring | |

| ID | Citation | Year | Genus | Species | Location | State | Transplant unit | Environment | Tide | Wave exposure | Experiment | Enhancement | Duration (years) | Success (%) | Objective | Depth |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Lord *et al.* 1999** | | | | | | | | | | | | | | | |
| 66 | **Meehan 2006 in Ganassin and Gibbs 2008** | | *Posidonia* | *australis* | St. Georges Basin | NSW | Sprig | estuary | subtidal | High | 1 | 1 | <1 | 10 | Research | 1m |
| 67 | **Roberts *et al.*? 2006 in Ganassin and Gibbs 2008** | | *Zostera* | *capricorni* | Tuggerah Lakes | NSW | Plug | estuary | inter and sub tidal | Sheltered | 1 | 0 | 1 | 100 | Research | |
| 68 | **Paling and Van Kuelen 2003 in Ganassin and Gibbs 2008** | **2003** | *Posidonia* | *australis* | Wagonga Inlet | NSW | Plug | estuary | subtidal | Sheltered | 1 | 0 | >1 | 92 | Creation trial | |
| 69 | **EP Consultin g Group 2004, 2007 in Ganassin and Gibbs 2008** | **2004; 2007** | *Zostera* | *capricorni* | Lady Robinson Beach | NSW | Plug | Marine | inter and sub tidal | High | 1 | 0 | 3 | | Research | |
| 70 | **Roberts *et al.* 2006 in Ganassin and Gibbs 2008** | **2006** | *Zostera* | *capricorni* | Foreshore Beach Botany Bay | NSW | | Marine | subtidal | Sheltered | 1 | 0 | 1 | | | |

| ID | **Citation** | **Year** | Genus | Species | Location | State | Transplant unit | Environment | Tide | Wave exposure | Experiment | Enhancement | Duration (years) | Success (%) | Objective | Depth |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 71 | **NSW Department of Primary Industries 2005 in Ganassin and Gibbs 2008** | **2005** | *Zostera* | *capricorni* | Yamba Bay Clarence River | NSW | Plug | estuary | inter and sub tidal | Sheltered | 1 | 0 | 1 | 0 | Research | |
| 72 | **Initial. Gordos *pers. comm.* 2006 in Ganassin and Gibbs 2008** | **2006** | *Zostera* | *capricorni* | Brunswick River | NSW | Plug | estuary | inter and sub tidal | Sheltered | 1 | 1 | 1 | 0 | Regeneration | |
| 73 | **Walker 1985 in Campbell 2010** | | *Amphibolis* | *spp.* | Cockburn sound | WA | Plug Seedlings | Marine | subtidal | Medium | 0 | 1 | | 50 | | |
| 74 | **Walker 1985 in Campbell 2010** | | *Amphibolis* | *spp.* | Warnbro sound | WA | Plug Seedlings | Marine | subtidal | Medium | 0 | 1 | | 50 | | |
| 75 | **Larkum 1976 in Campbell 2010** | | *Posidonia* | *australis* | Botany Bay | NSW | Plug | Marine | subtidal | Sheltered | 0 | | | | | |
| 76 | **LeProvost 1990 in Campbell 2010** | | *Posidonia* | *australis* | Cockburn sound | WA | Seedlings | Marine | subtidal | Medium | 0 | 1 | | 0 | | |

| ID | **Citation** | **Year** | Genus | Species | Location | State | Transplant unit | Environment | Tide | Wave exposure | Experiment | Enhancement | Duration (years) | Success (%) | Objective | Depth |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 77 | **Harris *et al.* 1980 in Campbell 2010** | | *Zostera* | *capricorni* | Illawarra Lake | NSW | | estuary | subtidal | Sheltered | 0 | | | | | |
| 78 | **Long 1990 in Campbell 2010** | | *Zostera* | *capricorni* | Cairns Harbour | QLD | Plug | Marine | subtidal | Sheltered | 0 | | | 100 | | |
| 79 | **Coles *et al.* 1990 in York and Smith 2013** | **1990** | *Zostera* | *muelleri* | Cairns Harbour | QLD | | Marine | subtidal | Sheltered | 0 | | <1 | | | |
| 80 | **Coles *et al.* 1990 in York and Smith 2013** | **1990** | *Thalassia* | *hemprichii* | Cairns Harbour | QLD | | Marine | subtidal | Sheltered | 0 | | <1 | | | |
| 81 | **Coles *et al.* 1990 in York and Smith 2013** | **1990** | *Halophila* | *decipiens* | Cairns Harbour | QLD | | Marine | intertidal | Sheltered | 0 | | <1 | 0 | | |
| 82 | **Coles *et al.* 1990 in York and Smith 2013** | **1990** | *Halophila* | *tricostata* | Cairns Harbour | QLD | | Marine | intertidal | Sheltered | 0 | | <1 | 0 | | |
| 83 | **Coles *et al.* 1990 in York and Smith 2013** | **1990** | *Zostera* | *muelleri* | Cairns Harbour | QLD | | Marine | subtidal | | 0 | | <1 | 79 | | |

| ID | **Citation** | **Year** | Genus | Species | Location | State | Transplant unit | Environment | Tide | Wave exposure | Experiment | Enhancement | Duration (years) | Success (%) | Objective | Depth |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 84 | **Poiner and Conacher 1992 in York and Smith 2013** | **1992** | *Zostera* | *muelleri* | Moreton Bay | QLD | | Marine | subtidal | Sheltered | 0 | | >1 | 0 | | |
| 85 | **Poiner and Conacher 1992 in York and Smith 2013** | **1992** | *Zostera* | *muelleri* | Moreton Bay | QLD | | Marine | subtidal | Sheltered | 0 | | 1 | | | |
| 86 | **McLennan and Sumpton 2005 in York and Smith 2013** | **1992** | *Zostera* | *muelleri* | Moreton Bay | QLD | Plug | Marine | subtidal | Sheltered | 0 | | <1 | 1 | | |
| 87 | **McLennan and Sumpton 2005 in York and Smith 2013** | **1992** | *Halophila* | *ovalis* | Moreton Bay | QLD | Plug | Marine | subtidal | Sheltered | 0 | | <1 | 1 | | |
| 88 | **Palingn *et al.* 2007** | **2007** | *Posidonia* | *sinuosa* | Cockburn Sound | WA | Plug | Marine | subtidal | Medium | 0 | | 2 | 0 | Research | 359m (transplant success decrease |

| ID | Citation | Year | Genus | Species | Location | State | Transplant unit | Environment | Tide | Wave exposure | Experiment | Enhancement | Duration (years) | Success (%) | Objective | Depth |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | | | | | | | d with depth) |
| 89 | **Paling *et al.* 2007** | **2007** | *Posidonia* | *sinuosa* | Cockburn Sound | WA | Sprig | Marine | subtidal | Medium | 0 | | 2 | 1 | Research | 359m |
| 90 | **Irving 2009; Irving *et al.* 2013; Tanner 2015** | **2009** | *Amphibolis* | *antarctica* | Largs Bay | SA | Sprig | Marine | subtidal | High/Medium | 1 | 1 | 1 | 1 | Research | 7-10m |
| 91 | **Irving 2009; Irving *et al.* 2013; Tanner 2016** | **2009** | *Amphibolis* | *antarctica* | Semaphore | SA | Sprig | Marine | subtidal | High/Medium | 1 | 1 | 1 | 1 | Research | 7-10m |
| 92 | **Irving 2009; Irving *et al.* 2013; Tanner 2017** | **2009** | *Amphibolis* | *antarctica* | Grange | SA | Sprig | Marine | subtidal | High/Medium | 1 | 1 | 1 | 1 | Research | 7-10m |
| 93 | **Irving 2009; Irving *et al.* 2013; Tanner 2018** | **2009** | *Amphibolis* | *antarctica* | Henley | SA | Sprig | Marine | subtidal | High/Medium | 1 | 1 | 1 | 1 | Research | 7-10m |
| 94 | **Irving *et al.* 2010** | **2010** | *Amphibolis* | *antarctica* | Henely Beach | SA | Plug | Marine | subtidal | Medium | 0 | 1 | <1 | 78 | Research | 6m |
| 95 | **Irving *et al.* 2010** | **2010** | *Amphibolis* | *antarctica* | Henely Beach | SA | Sprig | Marine | subtidal | Medium | 0 | 1 | <1 | 53 | Research | 6m |

| ID | **Citation** | **Year** | Genus | Species | Location | State | Transplant unit | Environment | Tide | Wave exposure | Experiment | Enhancement | Duration (years) | Success (%) | Objective | Depth |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 96 | **Irving *et al.* 2010** | **2010** | *Amphibolis* | *antarctica* | West Beach | SA | Plug | Marine | subtidal | High/Medium | 0 | 1 | <1 | 1 | Research | 3m |
| 97 | **Irving *et al.* 2010** | **2010** | *Amphibolis* | *antarctica* | West Beach | SA | Sprig | Marine | subtidal | Medium | 0 | 1 | <1 | 53 | Research | 3m |
| 98 | **Irving *et al.* 2010** | **2010** | *Heterozostera* | *nigricaulis* | Henely Beach | SA | Plug | Marine | subtidal | Medium | 0 | 1 | <1 | 78 | Research | 6m |
| 99 | **Irving *et al.* 2010** | **2010** | *Heterozostera* | *nigricaulis* | Henely Beach | SA | Sprig | Marine | subtidal | Medium | 0 | 1 | <1 | 53 | Research | 6m |
| 100 | **Irving *et al.* 2010** | **2010** | *Heterozostera* | *nigricaulis* | West Beach | SA | Plug | Marine | subtidal | Medium | 0 | 1 | <1 | 0 | Research | 3m |
| 101 | **Irving *et al.* 2010** | **2010** | *Heterozostera* | *nigricaulis* | West Beach | SA | Sprig | Marine | subtidal | Medium | 0 | 1 | <1 | 53 | Research | 3m |
| 102 | **Irving *et al.* 2010** | **2010** | *Posidonia* | *augustifolia* | Grange | SA | Seedlings | Marine | subtidal | High | 0 | 1 | <1 | 0 | Research | |
| 103 | **Irving *et al.* 2010** | **2010** | *Posidonia* | *augustifolia* | Semaphore | SA | Seedlings | Marine | subtidal | High | 0 | 1 | <1 | 0 | Research | |
| 104 | **Irving *et al.* 2010** | **2010** | *Posidonia* | *augustifolia* | Grange | SA | Seedlings | Marine | subtidal | High/Medium | 0 | 1 | <1 | 10 | Research | |
| 105 | **Irving *et al.* 2010** | **2010** | *Posidonia* | *augustifolia* | Semaphore | SA | Seedlings | Marine | subtidal | High/Medium | 0 | 1 | <1 | 10 | Research | |
| 106 | **Catchment to Coast Project in Tanner *et al.* 2014** | **2014** | *Posidonia* | *australis* | Kangaroo Island Western River Cove | SA | Sprig | Marine | subtidal | Sheltered | 0 | | 1 | 100 | Rehabilitation | <0.5m |
| 107 | **Catchment to Coast Project in Tanner *et al.* 2014** | **2014** | *Posidonia* | *australis* | Kangaroo Island Western River Cove | SA | Seeds | Marine | subtidal | Sheltered | 0 | | 1 | 0 | Rehabilitation | <0.5m |
| 108 | **Verduin and** | **2013** | *Posidonia* | *australis* | Cockburn Sound | WA | Sprig | Marine | subtidal | Medium | 1 | 1 | 3 | 100 | To restore seagrass following 77% | 2.2-4.0 |

| ID | **Citation** | **Year** | Genus | Species | Location | State | Transplant unit | Environment | Tide | Wave exposure | Experiment | Enhancement | Duration (years) | Success (%) | Objective | Depth |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Sinclair Year** | | | | (Southern Flats) | | | | | | | | | | reduction since 1967 largely due to the effects of eutrophication industrial develop-ment and sand mining | |
| 109 | **Western Port Seagrass Partnership Tim Ealey** | **2008** | *Heterozostera* | *tamanica* | Coronet Bay Western Port | Vic | Plug | Marine | intertidal | Sheltered | 1 | 1 | un-known | | | |
| 110 | **Western Port Seagrass Partnership Tim Ealey** | **2008** | *Zostera* | *muelleri* | Lang Lang | Vic | Plug | Marine | intertidal | Sheltered | 1 | 1 | un-known | 0 | | |
| 111 | **van Keulen *et al.* 2003** | **2003** | *Amphibolis* | *griffthii* | Carnac Island | WA | Plug | Marine | subtidal | High/Medium | 0 | 1 | 1 | 0 | examination of enhancement and unit size | |
| 112 | **Wear *et al.* 2010** | | *Amphibolis* | *antarctica* | Holdfast Bay | SA | | Marine | subtidal | High/Medium | 0 | | 1 | | | |
| 113 | **Statton and Dixon 2012** | **2013** | *Posidonia* | *australis* | Useless Loop Shark Bay | WA | | Marine | subtidal | Sheltered | 0 | | 1 | | | |
| 114 | **Bastyan and Cambridge 2008** | **2008** | *Posidonia* | *sinuosa* | Oyster Harbour | WA | Sprig | Marine | subtidal | High | 1 | 1 | >5 | 61 | Research | 5m |

| ID | **Citation** | **Year** | Genus | Species | Location | State | Transplant unit | Environment | Tide | Wave exposure | Experiment | Enhancement | Duration (years) | Success (%) | Objective | Depth |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 115 | **Statton and Dixon 2012** | **2013** | *Cymodocea* | *angustata* | Useless Loop Shark Bay | WA | | Marine | subtidal | Sheltered | 0 | | 1 | | | |
| 116 | **van Keulen *et al.* 2003** | **2003** | *Posidonia* | *australis* | Carnac Island | WA | Plug | Marine | subtidal | High/Medium | 0 | 1 | 1 | | examination of enhancement and unit size | |
| 117 | **Morrison 2009** | **2009** | *Posidonia* | *australis* | Shark Bay Useless Loop | WA | Plugs | Marine | subtidal | Sheltered | 0 | | 6 | 25 | Research | 4m |
| 118 | **Morrison 2009** | **2009** | *Amphibolis* | *antarctica* | Shark Bay, Useless Loop | WA | Plugs | Marine | subtidal | Sheltered | 0 | | 6 | 1 | Research | 4m |

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