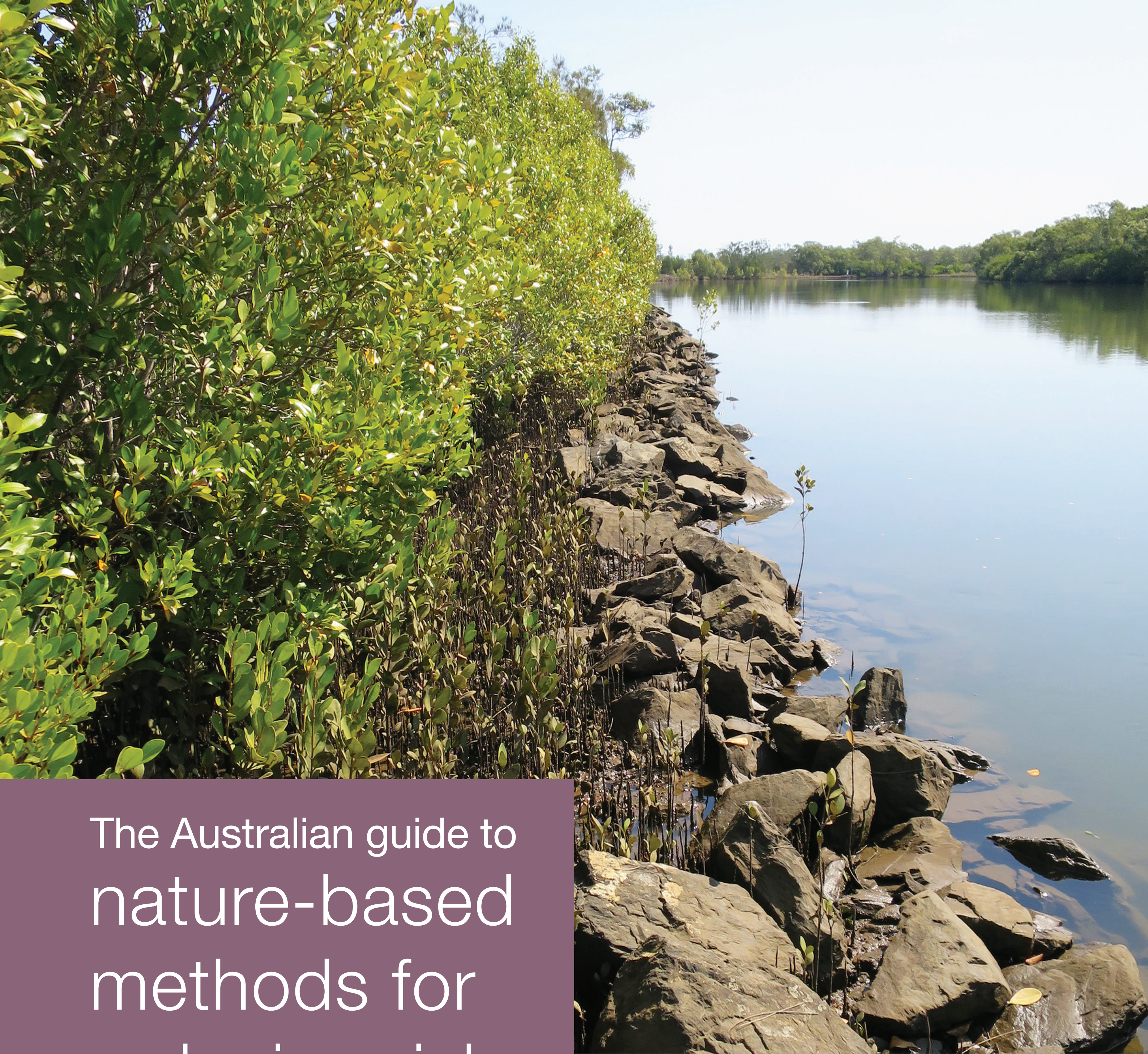

The Australian guide to nature-based methods for reducing risk from coastal hazards

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NCCC
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Author Contributions

The sections of this guideline were led by the following authors: coastal hazard risk reduction (AVR with MG and RJL); blue carbon (BJF); beaches (JTK); coastal dunes (TMK); saltmarshes (PB); mangroves (CEL); seagrasses (MWF with GAK); kelp forests (EMAS); shellfish reefs (MJB); coral reefs (NKB); benefit-cost analysis (AAR); policy framework (VS with EW and RLM); all other sections were led by RLM. The performance factors sub-sections were led/edited by AVR, MG and RJL. RLM and SES managed and edited the document. All authors contributed to discussions on the structure and commented on draft documents.

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Contents

Preface

The National Centre for Coasts and Climate (NCCC) was established at the University of Melbourne through the Earth Systems and Climate Change Hub of the Australian Government's National Environmental Science Program. The purpose of the NCCC is to work with stakeholders to identify the best ways of addressing climate change impacts in Australian coastal ecosystems. This has included research on the carbon storage of coastal vegetation, and historical and future impacts of coastal erosion. The knowledge has been applied to the development and trialling of nature-based methods for coastal hazard risk reduction to enhance the capacity of communities and ecosystems to adapt to climate change.

This publication has been developed by the NCCC in collaboration with researchers from six other Australian universities. The authors span expertise in the ecological, engineering, geomorphological, economical and socio-political aspects of nature-based methods.

This guide is directed at coastal managers and policy makers from local, state, and federal government, as well as engineers and other practitioners that work on the coast. It is designed to increase awareness of nature-based methods in Australia, and to outline what needs to be considered in their implementation. This guide is not intended to provide technical design guidance, but rather to introduce the suite of approaches that can be used. Technical design involving appropriate expertise should be sought prior to any project.

This report presents the first comprehensive guideline for nature-based methods specific to Australian systems. It complements existing guidelines by the National Committee on Coastal and Ocean Engineering (NCCOE) within Engineers Australia on climate change and adaptation, as well as the following international guidelines for nature-based methods:

National

- Guidelines for Responding to the Effects of Climate Change in Coastal and Ocean Engineering (NCCOE, 2012)
- Coastal Engineering Guidelines for Working with the Australian Coast in an Ecologically Sustainable Way (NCCOE, 2012)
- Climate Change Adaptation Guidelines in Coastal Management and Planning (NCCOE, 2012)

International

- Use of Natural and Nature-Based Features (NNBF) for Coastal Resilience (US Army Corps of Engineers, 2015)
- Implementing Nature-Based Flood Protection. Principles and Implementation Guidance (The World Bank, 2017)
- Guide for Applying Working with Nature to Navigation Infrastructure Projects (PIANC, 2018)
- Building with Nature. Creating, Implementing and Upscaling Nature-Based Solutions (EcoShape, 2020)

This is a living document that will be updated with new information. Coastal practitioners and other related professionals are encouraged to provide ongoing feedback and examples (to rebecca.morris@unimelb.edu.au).

Acknowledgements

Acknowledgment of Country

We acknowledge the traditional custodians of Country throughout Australia and recognise their continuing connection to land, waters and community. We pay our respects to ancestors and Elders, past, present and future. We recognise and acknowledge the contribution and interests of Aboriginal people and organisations in the management of coastal land and waters, and natural resources.

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We thank the 12 anonymous interview participants that informed the current policy landscape for nature-based methods in Australia (Section 5), conducted under the University of Melbourne human ethics permit # 2057435.1.

Figures

Graphics: Louise Bell (Lightbox 42)
Maps: Dean Chamberlain (University of Melbourne)

Image © Mitchell Shannon

Nature-Based Methods for Coastal Hazard Risk Reduction

Climate change and continued population growth are accelerating the need for diverse solutions to coastal protection. Traditionally shorelines are armoured with artificial, non-adaptive structures, which come with significant economic, environmental and social costs. While hard structures will continue to have a place in coastal protection, alternative methods that are more sustainable and climate-resilient should be more broadly adopted into the future where appropriate. Nature-based methods (through “soft” or “hybrid” techniques) have the potential to play important roles in climate adaptation and mitigation because of their ability to reduce the threats of coastal erosion and flooding and provide co-benefits such as carbon sequestration.

National Guidelines for Nature-Based Methods

Nature-based methods use the creation or restoration of coastal habitats for hazard risk reduction. This can be done through restoring the habitat alone (“soft” approach), or in combination with hard structures that support habitat establishment (“hybrid” approaches). The need to develop, test and apply more sustainable techniques to mitigate the impacts of coastal hazards has been identified as a national priority. One reason that nature-based methods have been underutilised in Australia is that decision-makers need clearer guidelines for when a soft, hybrid or hard coastal defence approach is most appropriate.

International exemplars in nature-based methods have started this process, which include Ecoshape’s Building with Nature in Europe and the Army Corps of Engineers’ Engineering-with-Nature® in the United States. Here we build on this international knowledge and national research efforts to provide an Australian context for nature-based methods, as wider adoption of these techniques nationally requires accounting for the environmental, economic and socio-political contexts specific to Australia.

This guideline summarises the physical processes that underpin nature-based methods, and the ecological and engineering considerations for their application based on the major coastal ecosystems found in Australia. It also provides frameworks for implementing nature-based methods and conducting a benefit-cost analysis, and the policy landscape within which nature-based methods can be applied. The aim of this document is to translate the known global and Australian research into a practical tool that can be used to support decisions by coastal practitioners to use nature-based methods.

Benefits of Nature-Based Methods

Natural ecosystems contribute coastal hazard risk reduction through ecosystem processes such as increased bed friction, local shallowing of water, sediment deposition and building of vertical biomass. These processes cause responses such as a change in shore profile and elevation relative to sea level, and wave attenuation, which in turn mitigate coastal hazards. As a living, growing system, nature-based methods are adaptive to a changing climate, and can self-repair after storm events. This is in contrast to traditional "hard" structures, which become less effective throughout their design life, and need to be upgraded or replaced with climate change. Shoreline hardening severs the transition between terrestrial and shallow marine ecosystems, resulting in a significant loss of biodiversity as natural habitats are replaced. Nature-based methods, on the other hand, have the capacity to provide a number of co-benefits in addition to coastal defence, which include supporting biodiversity, fisheries productivity, water filtration, and carbon storage.

Key Considerations for their Use

The environmental context and risk level are two key considerations in the decision to use a nature-based method. In particular, the habitat suitability and hazard exposure need to be appropriate for their use. In many instances these are interlinked (e.g., hydrodynamics can create a hazard and be a limiting factor for habitat establishment). In general, lower energy environments are more suitable for a soft approach, while hybrid approaches are more diverse and can be used in a larger range of environmental conditions. The risk level is lower where there is a larger buffer of land between the shoreline and infrastructure, and these areas are likely to be more suitable for nature-based methods; this is because they take time to grow. Nature-based methods also require space within the intertidal to shallow subtidal to be established, and a terrestrial buffer provides future adaptation space due to sea level rise. Where there is little land between the shoreline and infrastructure, increased hazards pose a high risk, and often it is too late to use a nature-based method. Where a nature-based method is suitable, there are a number of habitat-specific ecological requirements and engineering parameters that need be accounted for to achieve effective hazard risk reduction and habitat restoration (Table 1).

Barriers and Enablers for Nature-Based Methods

Despite the coastal policy landscape varying across jurisdictions, key coastal policy makers believe there is sufficient scope within existing policies to be able to apply nature-based methods. The key barriers to these approaches are timeframes and risk, funding, and precedent. Management and funding of risk reduction on the coast is predominantly directed towards areas that are at high risk. These areas are less likely to be suitable for nature-based methods, and there is pressure to choose tested options with a shorter timeframe to results. In areas that are suitable for nature-based methods, it is harder to justify forward-thinking management interventions in areas that are not at immediate risk. This reactive approach to management means that there is a lack of operational precedents of nature-based methods at scale showing what can be done in Australia. This lack of examples of standardised, best practice for nature-based methods means organisations may default to current practice regardless of the relevant policy context. An enabler of nature-based methods, however, is their capacity to provide a number of co-benefits, which means that their value can far outweigh their costs. Due to this, benefit-cost analyses that incorporate the market and non-market benefits of nature-based methods in comparison to traditional structures will be essential to provide a business case for the use of these interventions where suitable. This should include an analysis of the cost (and environmental) savings of employing a nature-based method now versus a hard solution in the future when the problem is exacerbated.

Table 1. A summary of the nature-based methods applicable to Australia. There are multiple approaches that can be applied within each ecosystem, which are described in detail in Section 2 of the guidelines.

	SANDY SYSTEMS		COASTAL VEGETATION				BIOGENIC REEFS	
	BEACHES	DUNES	SALTMARSHES	MANGROVES	SEAGRASSES	KELP FORESTS	SHELLFISH REEFS	CORAL REEFS
Key Process:								
Mechanism 1*	NA	NA	✓	✓	✓	✓	✓	✓
Mechanism 2*	NA	NA	×	×	×	×	✓	✓
Mechanism 3*	NA	NA	✓	✓	✓	✓	×	×
Co-benefits	Biodiversity; Fisheries; Water quality; CO_2 Carbon sequestration; Social value		Biodiversity; Fisheries; Water quality; CO_2 Carbon sequestration; Social value				Biodiversity; Fisheries; Water quality; Social value	
Factors affecting habitat suitability	Space; Sediment supply	Beach width; Sediment supply; Veg.	Temp.; Rainfall; Tidal elevation; Salinity; Waves	Temp.; Tidal elevation; Salinity; Waves; Sediment type, supply	Light; Salinity; Sediment type; Waves	Temp.; Light; Nutrients; Waves	Substrate; Temp.; Salinity; Dissolved O_2; Cl *a*, turbidity; Tidal elevation; Pollution	Temp.; Salinity; Waves/ Currents; Nutrient/ Sediment load; Light; Aragonite
Performance factors	Width; Volume; Grain size	Height; Width; Volume; Shape; Veg. cover, Density	Plant height; Bed width; Species; Density	Forest width; Species; Density	Plant height; Bed width; Species; Density	Plant height; Bed Width; Species; Density	Height; Width; Density of shellfish	Depth; Width; Coral morphology
Time to effectiveness†	•	•••	••••	••••	•••	•••	••	•••
Scalability‡	••	•••	•••	•••	•	••	•	••
Precedence§	•••	•••	••	•••	•	•	••	••
Construction costs (per m^2)¶	\$ - \$\$	\$ - \$\$\$	\$ - \$\$\$	\$ - \$\$\$	\$	\$ - \$\$	\$ - \$\$	\$ - \$\$
Maintenance required#	•••	•••	••	••	•••	••	•	••

* Mechanism 1: Wave attenuation due to roughness; Mechanism 2: Wave attenuation due to depth-induced breaking; Mechanism 3: Erosion mitigation within ecosystem; NA: not applicable, the key mechanism for sandy systems is to provide a physical buffer that erodes and recovers in response to storm events, see Section 2 of the guideline for more detail.

Co-benefits: Biodiversity; Fisheries; Water quality; CO_2 Carbon sequestration; Social value

† In years: • ≤1; •• ≤ 1-5; ••• ≤ 1-10; •••• ≤ 1-25

‡ Relevant to scales > 1 km^2: ••• All interventions are scalable; •• Most interventions are scalable; • Some interventions are scalable

§ Precedence for risk reduction: ••• Yes; •• Some; • None

¶ In AU\$ per m^2: \$ <100; \$\$ 100 – 500; \$\$\$ 500 – 1000. The average cost per linear metre of revetment is \$1700 and seawall is \$2100.

\# Number of interventions needing maintenance: ••• All interventions need maintenance; •• Most interventions need maintenance; • Some interventions need maintenance

1 An Introduction to Nature-Based Methods

Climate change is projected to increase the risk of coastal hazards driven by accelerating sea-level rise, a changing wave climate, and potentially more intense or frequent storm events[1, 2]. In Australia, the greatest population growth is occurring at the coast[3], which is increasing the exposure of people to current and future coastal hazards. These combined, are increasing the pressure on investment in coastal protection infrastructure. Hard engineered structures, such as seawalls and rock revetments, have been a common solution to coastal risk reduction in Australia. However, these static structures have little capacity to adapt to changes in climate, are expensive to build and maintain, and have serious socio-ecological consequences as natural functioning shorelines are replaced with artificial ones[4]. This has increased the interest in alternative approaches, which include nature-based coastal defences, within a suite of other adaptation options (Figure 1.1).

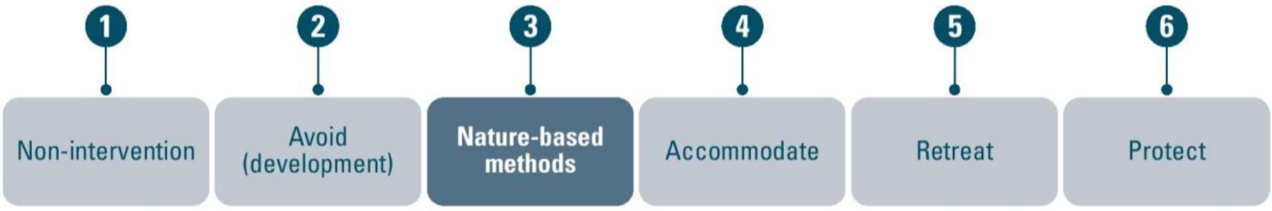


Figure 1.1: Adaptation Pathway Approach for managing coastal hazard risk, with actions presented in order of consideration (from left to right; adapted from the Victorian Marine and Coastal Policy 2020).

1.1 Definition and Document Scope

Nature-based methods, also referred to as 'nature-based coastal defence[5] or a 'living shoreline[6], is the creation or restoration of coastal habitats for hazard risk reduction. This includes the rehabilitation of existing degraded habitats, restoration of those historically present, or the creation of new habitats in ecologically suitable areas (hereafter collectively referred to as 'restored'). Typical habitats included in nature-based coastal defences are beaches and dunes, saltmarshes, mangroves, seagrasses and kelp forests, coral and shellfish reefs, alone or in combination. Nature-based defences can restore the habitat alone ("soft" approach), or in combination with hard structures that support habitat establishment ("hybrid" approaches). The key aim of nature-based coastal defence is to restore the ecological processes and functions that underpin the delivery of the natural coastal defence service.

The suitability of the environment to establish coastal habitats, as well as the severity of the hazards, and risk to the assets (of built, social or natural value) will determine the type and success (i.e. appropriateness) of a nature-based method. While traditional structures have a long history of use, guidance around when and where to use nature-based coastal defences in Australia is lacking. The purpose of this document is to provide clearer guidelines to decision-makers for when a soft, hybrid or hard approach is most appropriate, and the process for their implementation.

Section 1

Outlines how nature-based coastal defences work, the benefits and co-benefits, and broad considerations for their use.

Section 2

Details the ecological and engineering requirements of each habitat used in nature-based methods.

Section 3

Provides a framework for implementing nature-based coastal defences.

Section 4

Provides a framework for conducting a benefit-cost analysis to evaluate different methods.

Section 5

Outlines the policy framework relevant to nature-based coastal defence.

A soft approach: saltmarsh with substrate for oyster reefs © Rebecca Morris

A hybrid approach: rock sill with saltmarsh © Rebecca Morris

1.2 Coastal Hazard Risk Reduction

This section provides an overview of the oceanic and atmospheric processes that are responsible for coastal hazards globally, including Australia (by driving flooding and erosion) and the mechanisms by which coastal ecosystems can mitigate these hazards.

Coastal Hazards

Flooding

While flooding in coastal regions can be driven from the landward side by extremes in river discharge during storms, flooding along most coastlines is driven by extremes in the total water level generated within the ocean. The nearshore total water level is the sum of the offshore still water level (driven by e.g., tides, seasonal and inter-annual variations and long-term sea level rise), atmospheric surge, as well as wave-driven processes such as wave setup and swash motions (Figure 1.2). The motions that drive the water level variability occur over characteristic time-scales, including (from shortest to longest): wind-generated sea-swell waves (1 – 25 seconds), long-period (infra-) gravity waves (25 seconds – tens of minutes), (astronomical) tides (hours – years), atmospheric surges (hours – days), coastally trapped waves (10 – 25 days)[7], seasonal and inter-annual variations (months – years), and long-term (> decades) mean sea level rise, as well as episodic events (e.g., tsunamis with periods < 1 hour)[8]. Extremes in total water level at the shoreline that cause flooding are typically induced by interactions between multiple physical processes acting in concert (including indirect processes such as wave setup), rather than one single process alone (Figure 1.2). Although a single process can dominate in some areas, for example sea level rise is a major driver of flood exposure in estuaries[9].

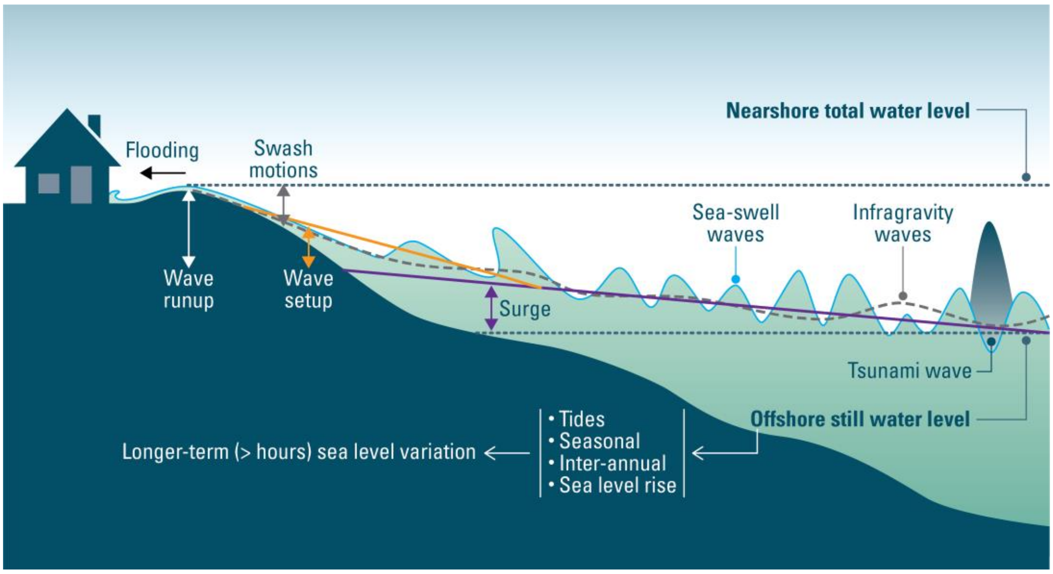


Figure 1.2: Contributions to the nearshore total water level that determine the potential for coastal flooding.

Wave runup (wave setup and swash motions)

Wind-generated waves are typically classified as either sea or swell waves and represent the dominant form of (non-tidal) wave energy incident to coastlines. 'Sea waves' (1 – 5 second period) are generated locally while 'swell waves' (5 – 25 second period) are generated by winds at distant locations and travel towards the coast. The forces that are generated by wave breaking of sea-swell waves cause a mean increase in the water level shoreward of breaking called 'wave setup'[10] (Figure 1.2). Sea-swell waves typically travel in wave groups of similar wave height, which leads to the generation of 'infragravity waves', especially when wave groups break in the nearshore (Figure 1.2). Off the coast, infragravity waves have much lower amplitudes and longer periods (< 25 s) than shorter sea-swell waves; however, they can increase in height and become important contributors to the total nearshore water level during storms and along coastlines fronted by reef systems[11]. Both sea-swell and infragravity waves cause oscillations in the water level at the coastline due to 'swash motions'. The total wave-induced vertical excursion of the water level ('wave runup') equals the sum of the steady (time-averaged) wave setup and the instantaneous swash motions (Figure 1.2). Wave runup tends to be most pronounced along unprotected coastlines on narrow continental shelf regions where offshore waves reach coastal regions without a major loss of energy[12] and thus is more significant on beaches in the SE, S and SW of Australia. Local beach slope also affects wave runup with significantly higher runup on steep beaches compared with flat ones[13].

Storm surge

Wind- and pressure-driven increase in the mean water level ('storm surge') is caused by either changes in air pressure (barometric setup) or by wind blowing across the ocean surface and pushing water up against the coast. Although usually small most of the time, storm surge can become a substantial contribution to the total water level during strong winds, particularly on shallow, wide continental shelfs or (semi-)enclosed systems such as tidal basins[14] and thus is more significant in Northern Australia.

Offshore still water level

The offshore still water level is defined as the water level averaged over time-scales larger than sea-swell waves, and is driven by longer-term (> hours) variations due to e.g., tides, seasonal and inter-annual variability and sea level rise (Figure 1.2). While most (open) coastlines around the world experience a tidal cycle with two high and two low waters per (lunar) day ('semi-diurnal tide'), the presence of the large continents on Earth causes some coastal areas to experience only one high and low ('diurnal tide') or experience two high and low tides of different size ('mixed semi-diurnal tide') daily. In Australia, the east coast and northern Western Australian coastline (Kimberley region) are generally characterised by semi-diurnal tides, while the south-west coast facing the Indian Ocean and the Gulf of Carpentaria experiences diurnal tides. The remaining coastal areas around Australia generally experience mixed tides. Although the daily tidal variation is the dominant contributor, longer-term tidal contributions exist such as 'spring tides' (caused by the near alignment of the Earth, sun and moon during full or new moons where the gravitational pull is at its maximum, leading to a larger tidal range) and 'neap tides' (due to the sun and moon being at right angles to each other during the first and third quarter moon leading to minimum gravitational pull and lower tidal range), which occur twice per month. On even longer timescales, interannual tidal modulations, in particular due to the 18.61 year lunar node cycle and the 8.85 year cycle of lunar perigee (which acts as a quasi 4.4 year cycle), can be substantial with amplitudes up to several decimetres[15]. Although the tidal dynamics vary greatly across Australia (Figure 1.3a) and can become substantially more complex in tidal inlets and estuaries that may see amplification or attenuation depending on estuarine type[9], tidal water levels are generally highly predictable, contrary to the other contributions to the total water level.

Coastally trapped waves form along sloping seabeds under alongshore wind stress and can travel over long distances (> 1000 km) before being dissipated due to bottom friction. In Australia, they occur particularly along the SW, S, and E coast and can contribute substantially to the total water level with periods of 10 to 25 days, wave lengths of several kilometres, and amplitudes varying from several centimetres to decimetres depending on the width of the continental shelf as well as the season[7].

At longer time-scales, seasonal and inter-annual/decadal sources of sea level variability can be substantial along many coastlines worldwide, including playing an important role in coastal flooding in many parts of Australia. Seasonal variations in water level in Australia (in the order of 10 cm) are largely driven by variations in the two major boundary current systems (the East Australian Current and Leeuwin Current). Inter-annual and decadal variations are the result of similar regional ocean processes and tend to be strongly connected to global climate cycles, such as the El Nino-Southern Oscillation (ENSO), Indian Ocean Dipole (IOD), Pacific Decadal Oscillation (PDO), and the Interdecadal Pacific Oscillation (IPO). Over even longer time-scales, mean sea level rise is having an increasing influence on sea level changes, with an average rising rate of 3.2 mm per year around Australia since 1993, with local variations ranging between 1 and 11 mm per year (Figure 1.3b[16]). The global average rise in sea level by 2100 is projected to be 0.29 to 0.59 metres for the low emissions scenario and 0.61 to 1.1 m for the high emissions scenario relative to 1986– 2005[17].

Erosion

Coasts are dynamic and undergo cycles of erosion and accretion that vary on many time scales. A distinction is made between long term (occurring over years or more) and short term (occurring over hours to days) erosion. Long term erosion causes the shoreline to retreat over time and is usually the result of a change in ocean conditions (e.g., waves, sea level), changes to sediment supply (from land or the ocean), and/or human interventions at the coastline. Short term erosion mainly occurs during storms (sometimes occurring in a cluster) and typically results in a temporary change, for example a beach and dune system with reduced width and volume. Over time, a sedimentary coastal system is often able to restore itself naturally through sediment transport driven by a combination of wind, waves and ocean currents. In Australia, erosion is most publicised when it occurs at beach and dune systems adjacent to coastal settlements. However, erosion of other sedimentary shorelines such as mudflats, mangroves and saltmarshes can also occur in response to these same drivers. Along a natural sedimentary shoreline, erosion volumes depend not only on nearshore ocean conditions, but also the sediment grain size and the steepness of the shore. Infrastructure built on or near sedimentary ecosystems may also influence erosion rates substantially; for instance, by limiting the alongshore transport of sediment or enhancing local erosion in the vicinity of the structure (scour). Globally, many coastlines suffer from ongoing erosion due to reduced river sediment supply, such as when dams are constructed upstream[18]. This is less of an issue for Australia, although projected future changes in local climate may lead to changes in river flows, which in turn may affect sediment input into the coastal system[19].

Generally, all sandy coastlines in Australia are at risk of erosion in the future as a result of sea level rise and changing wave climate conditions. A recent global analysis projects shoreline retreat up to 200 m by 2100 for a large number of beaches around Australia for a high emissions scenario[20], although their methods have been subject to discussion for being too simplified[21]. In particular for Australia, with its complex coastline with reefs, islands, inlets, and an abundance of nearshore marine ecosystems, predicting the coastal evolution over such timescales is challenging. However, with sea level expected to rise and the lack of accommodation space on the land side due to coastal infrastructure, it is evident coastal erosion challenges are likely to increase, particularly along urbanised coastlines near the capital cities. Many local governments are currently already forced to apply mitigation measures for beaches that suffer from substantial erosion. These areas are often referred to as 'erosion hotspots' and can be found across the country (some examples are shown in Figure 1.3f). Collectively, they cannot be directly linked to a single coastal hazard component alone (Figure 1.3a-e).

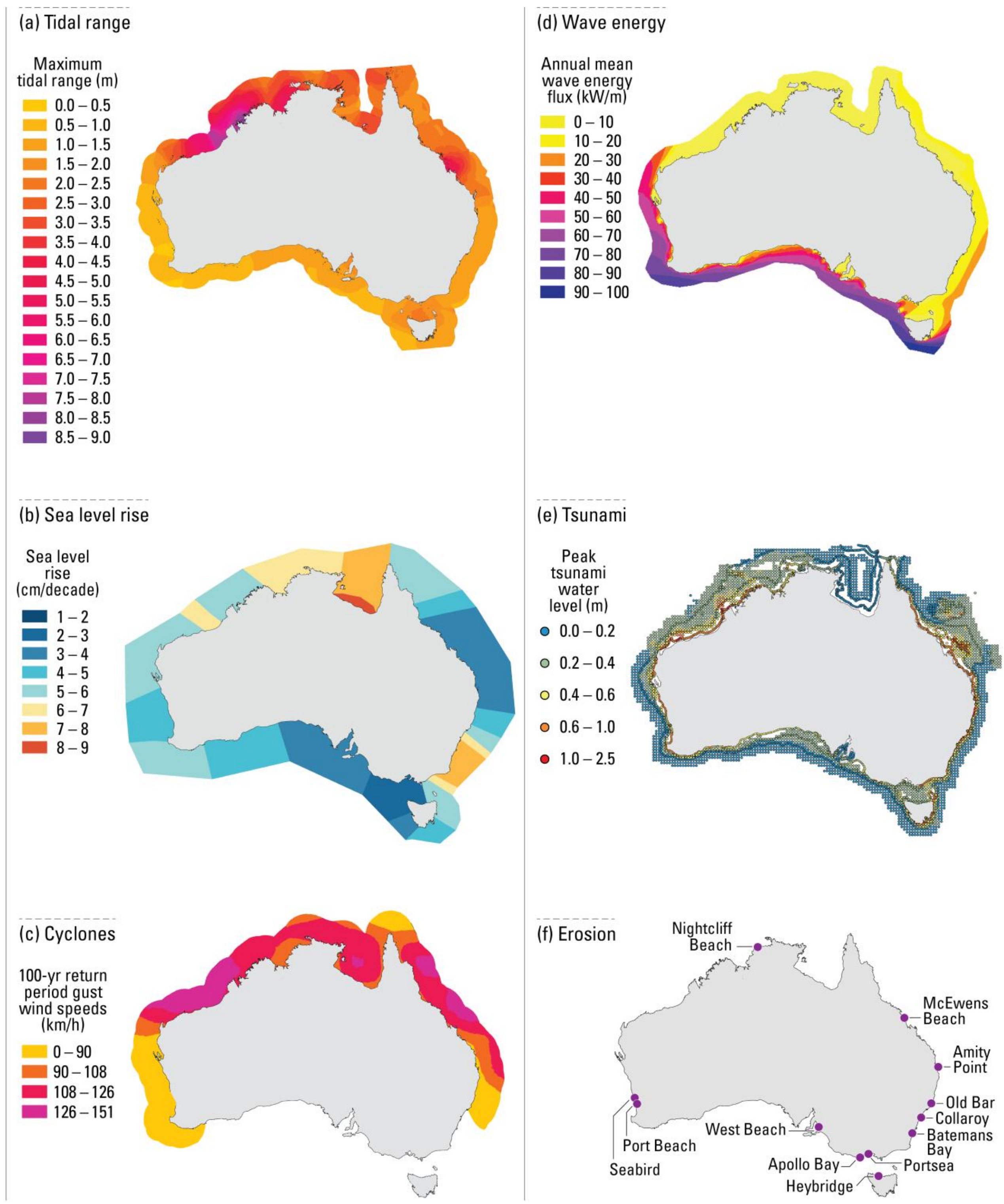


Figure 1.3: Distribution of drivers of coastal hazards across Australia. a) Average tidal range at spring tide; b) Average mean sea level rise since 1993; c) 100-year return period gust wind speeds due to tropical cyclones; d) Annual mean wave energy flux; e) 500-year return period for the maximum tsunami flood level; f) some examples of recent or current erosion hotspots (note this is not a comprehensive list).

Wave and Setup Attenuation by Ecosystem Roughness

Marine ecosystems that include beaches and dunes, saltmarshes, mangroves, seagrasses, shellfish and coral reefs and kelp forests are characterised by having large assemblages of individual organisms that create large-scale ecosystem roughness (often termed 'canopies'), much like forest canopies on land. This roughness created in the coastal zone can cause substantial dissipation of wave energy, caused by drag forces generated as the flow interacts with the roughness (Figure 1.4).

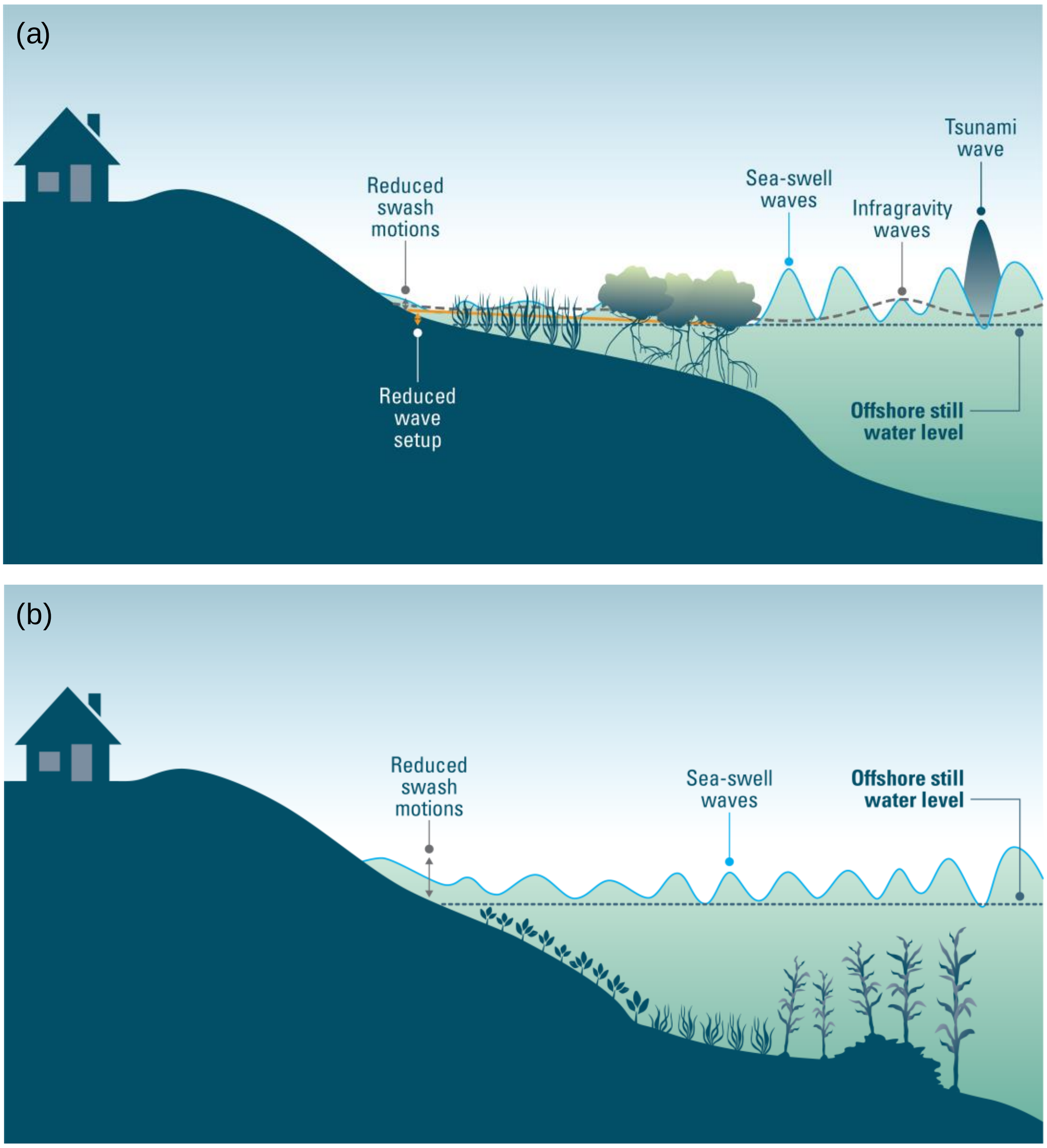


Figure 1.4: Coastal hazard mitigation through ecosystem roughness for emergent (salt marsh, mangrove; typical of estuarine settings), and submerged (e.g., seagrass, kelp) marine ecosystems. a) Emergent ecosystems may cause substantial attenuation of sea-swell waves, infragravity waves, wave setup and tsunami waves; b) The presence of submerged ecosystems can also lead to substantial attenuation of sea-swell waves, although these systems are generally less effective relative to emergent ecosystems as they only interact with the waves across a relatively small part of the water depth.

The rate of energy dissipation is dependent on the incident wave conditions, organism shape and flexibility as well as density per unit area and areal coverage[22]. In general, marine ecosystems that form emergent canopies that extend over the full water column at a given time point (e.g., mangroves, salt marshes) tend to be most efficient in wave damping. These species tend to be relatively rigid and by interacting with the waves across the entire water depth, drag forces are relatively large, which results in high rates of wave energy dissipation. Intertidal marine ecosystems that become fully submerged at high tide (e.g., shellfish reefs, some seagrasses and kelps), or subtidal ecosystems (e.g., corals, some seagrass, kelp and shellfish species) occupying only a fraction of the water column can be less effective if they are at a water depth where they only interact with waves across a relatively small part of the water column (Figure 1.4). However, where these ecosystems are very shallow and/or have a substantial ground footprint, their wave attenuation capacity can still be high[23, 24]. Although energy dissipation due to ecosystem roughness is primarily applicable to sea-swell waves and their resulting swash motions, attenuation of infragravity waves may be substantial for ecosystems with relatively large spatial coverage[25]. In cases of emergent vegetation and/or relatively long waves relative to the water depth, a mean drag due to ecosystem roughness causes a reduction in wave setup[26]. Lastly, emergent vegetation, in particular mangroves, can also substantially attenuate tsunami waves[27].

Wave Attenuation by Depth Induced Wave Breaking

By limiting the water depth, biogenic reefs (e.g., corals and shellfish) can dissipate substantial wave energy from incoming sea and swell waves. Where reefs are present, wave breaking is induced further offshore compared to when reefs are not present, reducing the amount of sea-swell wave energy reaching a coastline in the lee of the reef. During wave breaking, part of the energy is transferred into an increase in mean water level on the reef (wave setup) as well as infragravity wave motions, which can both become relatively important contributors to the nearshore water level (Figure 1.5).

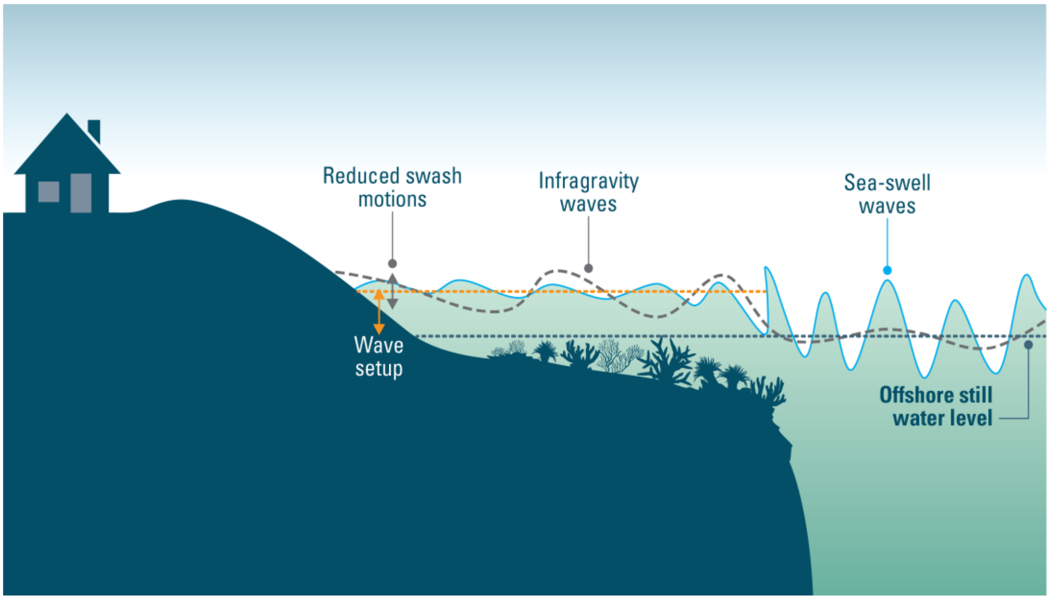


Figure 1.5: Coastal hazard mitigation through wave attenuation by depth-induced wave breaking by biogenic reefs (e.g., corals and shellfish).

The process of sea-swell wave energy reduction through breaking is, however, generally considered dominant, resulting in reduced wave runup levels. The effectiveness of reefs in dissipating wave energy through breaking depends on their location and height relative to the water surface elevation. Waves generally break when their height is greater than about three quarters of the water depth[28] and for most reefs, energy reduction due to breaking is considered dominant over friction. Nearshore sand bars that are generated by large volumes of sand eroded from the beach and dunes during severe storms act in a similar way to reefs by reducing the water depth, thereby increasing wave breaking farther offshore and reducing impact on the shoreline during the remainder of the storm. In conventional coastal engineering practice, submerged breakwaters utilise the same concept to limit the wave impact on the shoreline.

Storm Surge Reduction by Marine Ecosystems

Although the number of studies focusing on how ecosystems influence storm surge is relatively small, there is broad consensus that certain marine ecosystems, in particular shallow regions with salt marshes and mangroves, can be effective in limiting maximum nearshore water levels due to wind-generated surge[29]. The drag exerted by these ecosystems may slow down the flows that are needed to build up the surge, leading to a spatial pressure gradient with lower onshore water levels relative to offshore locations (Figure 1.6). Ecosystems with greater biomass (density, height, volume) are expected to have a greater effect due to increased drag, while spatial discontinuities such the presence of pools and channels are considered to decrease the potential for surge reduction as they allow for water volumes to flow more freely[30]. The drag is also dependent on the flow velocity and thus the potential for surge reduction becomes larger with increasing storm intensity[31]. It is important to note, however, that under continuous wind forcing the reduction of surge by ecosystems decreases over time and eventually becomes negligible over longer durations as the nearshore water level is able to catch up with the offshore water level. Consequently, ecosystem drag is considered most effective for relatively large, fast-moving storm systems[29].

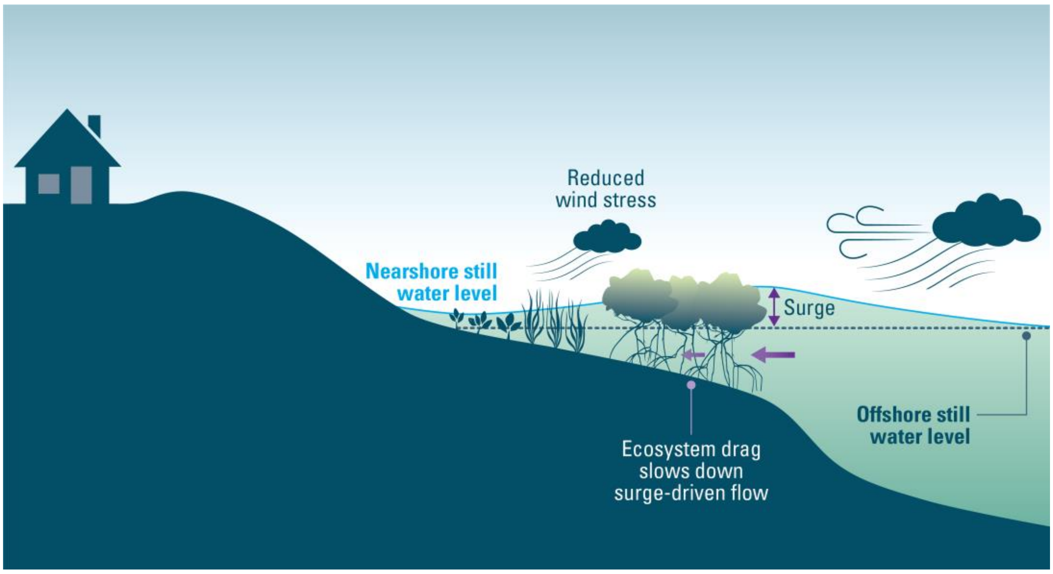


Figure 1.6: Reduction in maximum storm surge levels by mangroves and saltmarsh through slowing of the storm surge and a reduction in local wind stress.

Besides the lowering of maximum water levels by slowing down the surge, emergent vegetation such as salt marsh and mangroves may also greatly reduce local wind stress acting on the water surface (Figure 1.6), which has been found to lead to local reductions in maximum nearshore water level of over one metre in a U.S. hurricane modelling study[32].

Erosion Mitigation by Natural Systems

In addition to the capacity of marine ecosystems to reduce the impact of waves and water levels on the coastline, they may also promote a stable coastline more directly by influencing sediment transport processes (Figure 1.7). The canopies formed by coastal ecosystems can create reduce near-bed flows and bed shear stresses, which facilitate sediment settling and reduced erosion[33]. Both marine and dune vegetation can also help limit erosion by binding the sediment with their roots. Dune vegetation helps trap wind-blown sand and is critical to maintain a sediment buffer that is available to be transported offshore as sand bars adjust to higher water levels and waves during storms. This can help to limit the extent of erosion. Many organisms within coastal ecosystems also produce calcium carbonate skeletons (e.g., corals, shellfish) that, when broken down, form sediments that help build up the beach over longer timescales. In conventional engineering, erosion is generally mitigated through bottom protection mats to prevent waves and currents from suspending sediment.

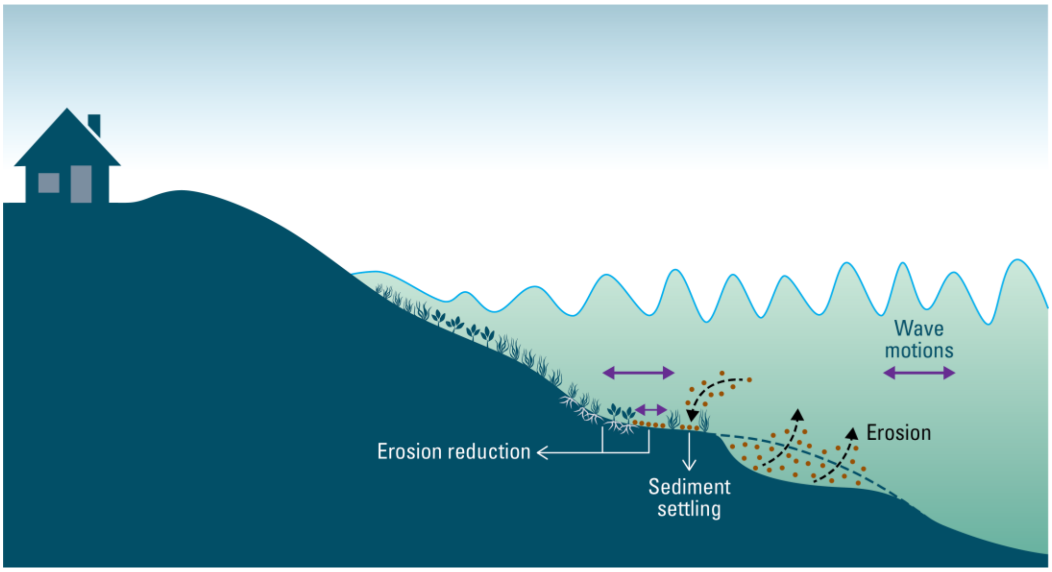


Figure 1.7: Marine ecosystems can enhance coastal stability by 1) stabilising the sediment (and limiting the potential for erosion) through their roots, and 2) reducing the near-bed flow velocity, thereby promoting sediment deposition.

Ecosystem Response to Extreme Conditions

Although marine ecosystems can be effective in providing coastal protection, there are still limited observations under extreme conditions, where there is the potential for the protection provided to be diminished when the ocean forcing is simply too great and is able to deform (e.g., bending over of flexible seagrass[34]) or even damage the ecosystem itself (e.g., by uprooting vegetation or damaging corals[35]). While decades of research and practice have resulted in well-established guidelines and regulations for conventional engineering structures, this is not the case for the complex and diverse range of nature-based methods. The threshold of a marine ecosystem to provide effective coastal protection under extreme conditions is dependent on a range of variables including species, growth stage, spatial coverage and the incident wave conditions. However, contrary to conventional structures, marine ecosystems may be able to self-repair damage from a severe storm if there is sufficient time before the next storm event occurs[36]. This is one of the potential benefits of nature-based over traditional coastal protection (see Section 1.3: Benefits of Nature-Based Methods).

Summary of Ecosystem Reduction of Coastal Hazard Risk

As described in the previous sections, the efficacy of ecosystems at reducing coastal hazard risk is dependent on both ecological and environmental characteristics, which are summarised in Table 1.1, and described in more detail for each of the habitats in Section 2.

Table 2.1 The physical mechanisms through which the characteristics of coastal ecosystems are linked to hazard risk reduction.

	CHARACTERISTIC	MECHANISM 1: WAVE ATTENUATION DUE TO ROUGHNESS	MECHANISM 2: WAVE ATTENUATION BY DEPTH-INDUCED BREAKING	MECHANISM 3: EROSION MITIGATION WITHIN ECOSYSTEM
ECOSYSTEM	**Cross-shore width**	+	0	+
	Ecosystem height, relative to water depth	+	+	+
	Density/coverage	+	0	+
	Morphology: frontal area per unit biomass	+	0	+
	Root/rhizome density	0	0	+
	Flexibility	-	0	-
LOCAL	**Local depth**	-	-	-
	Wave height	+	+	+
	Wave period	-	-	+

+ = anticipated positive correlation between mitigation of coastal hazard and this variable
- = anticipated negative correlation between mitigation of coastal hazard and this variable
0 = no anticipated effect

1.3 Benefits of Nature-Based Methods

Nature-based coastal defences can have direct benefits over traditional structures in terms of their use for coastal hazard risk reduction. They also have co-benefits that are an outcome of the rehabilitation of natural habitats (Figure 1.8).

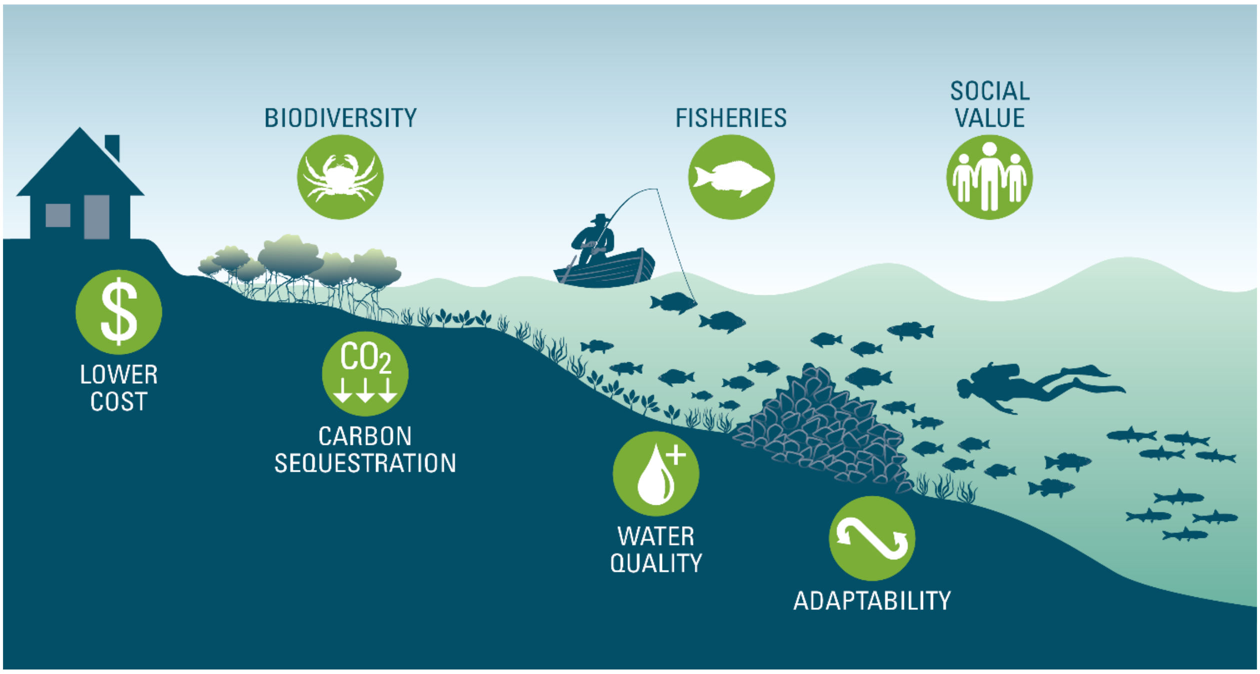


Figure 1.8: Benefits and co-benefits of nature-based methods.

Benefits

Adaptability

While traditional structures are largely static, meaning that they need to be upgraded or rebuilt in response to a changing climate, nature-based coastal defences have the capacity to adapt. In response to sea level rise, coastal habitats can adapt in one of two primary ways: (1) vertical accretion or growth; or (2) migration landwards across low-lying areas. Current evidence suggests that mangrove areas[37] and coral[38] and shellfish[39] reefs can accrete at a rate that matches sea level rise. Shallow seagrasses may also be unaffected by sea level rise, if they can expand into newly inundated areas[40]. Although within Australia, there is a current pattern of mangrove encroachment into *Melaleuca*, *Casuarina* or saltmarsh-dominated areas due to saltwater intrusion, and lower accretion rates in saltmarsh[41]. Once habitats can no longer accrete to keep up with sea level rise, they will need to have space to retreat. Landward migration of beaches and dunes is predicted in response to sea level rise[42]. This is an important consideration for the future resilience of nature-based coastal defences (Section 2 – Designing for the future).

Lower Maintenance Costs

Traditional structures that become damaged during a storm event need to be repaired, which are often funded in Australia through local or state governments, or the federal government's National Disaster Relief and Recovery measures[43]. In contrast, nature-based coastal defences can self-repair as they are a living, growing system. In the United States living shorelines using saltmarshes suffered less hurricane damage compared to bulkheads, and repaired themselves within one year with no reduction in shoreline elevation[36]. In Australia, defoliated mangroves can recover following cyclones[44], and die-backs in response to these are patchy and expected to be counteracted by the landward expansion observed across Australia[45]. The recovery time of these systems, as well as for seagrasses and coral reefs[46], depends on the magnitude of damage, as well as other management strategies (e.g., controlling pest species, addressing water quality issues) that increase habitat resilience.

Higher Benefit-Cost Ratios

Nature-based coastal defences can be considerably cheaper to construct than traditional structures such as seawalls[23, 47, 48] (see Section 2 for specific costs). For example, in a cost analysis of different options for erosion risk reduction in Western Port Bay, Victoria even the most expensive hybrid mangrove option was a third of the cost of a rock sill, and 3.5 times cheaper than a seawall or revetment[48]. Therefore, where it is appropriate to use nature-based methods, there is often an upfront cost saving. Even if the cost of a nature-based method is similar to a traditional one, as nature-based coastal defences incorporate the restoration of coastal habitats, they have a number of other co-benefits of economic importance. These co-benefits are not achieved with traditional structures and can often result in the benefits of nature-based methods far outweighing the costs (Section 4 – Cost-benefit analysis for nature-based defences).

Co-benefits

Creation and Preservation of Habitat and Biodiversity

Nature-based coastal defences use key habitat-forming species, and their restoration can support a diverse suite of marine organisms that use those habitats for shelter and/or food[49, 50]. It has been well-documented that nature-based methods support a higher abundance and diversity of fauna compared to traditional structures[51]. However, depending on how the habitat is engineered for coastal defence, there may be some ecological trade-offs[52]. For example, where rock sills are used in combination with vegetation in a hybrid approach, filter-feeding organisms can colonise the rock sills that would not be present in a natural vegetated area without hard substrate[52]. This means that the ecological niche provided by nature-based methods can differ from a natural system[53].

Climate Mitigation through Carbon Sequestration

Mangroves, saltmarshes, and seagrasses (Blue Carbon Ecosystems or BCEs) are internationally recognised for their ability to sequester carbon in their above and below ground biomass; and to trap, bury, and store carbon in their sediments[54]. Carbon in BCE sediments can be stored on century to millennial timescales because the anoxic and saline sediment conditions in BCEs reduce and inhibit metabolic decomposition pathways that would otherwise lead to re-emission of the stored carbon as methane and carbon dioxide. Protection, restoration, and creation of BCEs can mitigate climate change through maintaining/increasing their existing carbon stocks and carbon sequestration capacity along with avoiding carbon emissions that result from BCE degradation[55]. This includes via nature-based methods that integrate one or more BCEs into their design, which could sustain and/or increase carbon stocks and carbon sequestration capacity compared to hard engineered solutions. Given national and international efforts to include BCE restoration as an eligible method for attracting carbon credits in carbon trading schemes (e.g., Australia's Emission Reduction Fund[56]), there are also opportunities to offset some of the construction and maintenance costs of nature-based coastal defences via carbon credits. However, the magnitude of this co-financing option will strongly depend on the extent of the area under consideration, and on how many different BCEs can be successfully integrated into a nature-based coastal defence approach.

Maintenance of Fisheries

Coastal habitats play a significant role in supporting the survival and growth of commercially and recreationally important fish and invertebrate species. For example, coastal wetlands in south-east Australia have been valued at $35.6 million annually for their contribution to coastal fisheries[57]. These habitats provide a complex mosaic among which fish migrate to reproduce and forage and seek shelter as smaller juveniles. Nature-based coastal defences have the potential to contribute to this service by expanding areas of foraging habitat and nursery grounds for juvenile stages within a seascape. They can support higher abundances of commercially-important species compared to areas without nature-based defences, those traditionally armoured, and even natural shorelines in some cases[51, 58].

Improvement in Water Quality

Coastal vegetation, such as mangroves and macroalgae can absorb excess nutrients and organic load, as well as bioaccumulate heavy metals. These habitats have been used previously as a tool for wastewater treatment[59]. Shellfish feed by filtering organic particles from the water column, processing vast volumes of water each day. For example, the Chesapeake Bay, United States was filtered every three days by historical oyster reefs[60]. Shellfish have therefore been the target of management programs to improve water quality[61]. This ability for nature-based coastal defences to be biofilters is valuable on urbanised coastlines, which often have high levels of water and sediment pollution.

Social Value

Natural shorelines are highly valued by the public for important cultural (e.g., spiritual and religious beliefs and heritage value), direct-use (e.g., recreation, tourism, education) and behavioural (e.g., human well-being) services. Indigenous Australians have had a connection with their "Sea Country" for tens of thousands of years, sustainably managing coastal, marine and island resources[62]. Recreational use of the coast is often centred around those areas that have greater natural value, for example birdwatchers and fishers will travel to extensive areas of saltmarsh and seagrass as they have a greater abundance of target species[57]. Studies have shown that local residents are concerned about the coastal environment, and there is strong support for adaptive management that is ecologically sensitive[63].

1.4 Considerations for their Use

The *environmental context* and the *risk level* are the two key components that should inform the technical selection of the intervention. This decision includes firstly whether nature-based methods (i.e., soft or hybrid) are appropriate, and secondly what design should be used (i.e., the particular habitat/s, species). This section considers more generally the decision to use a nature-based method, while Section 2 (Ecological and engineering considerations for nature-based defences) considers the details of each habitat specifically.

Environmental Context

The hazard intensity and habitat suitability need to be appropriate for the use of a nature-based method. Broadly, for example, dunes and beaches are common features of open sandy coasts, whereas saltmarshes, mangroves, and shellfish reefs are found in comparatively sheltered estuaries and bays. These habitats alone can provide effective hazard risk reduction[36, 64]. However, under higher-intensity hazards soft approaches are likely to require more maintenance, as the natural recovery processes can takes months to years depending on the frequency of recurrence of hazard events, as well as processes such as sediment volume and supply, and recruitment of new individuals[64, 65]. Where the historical clearing of habitats has resulted in an increase in hydrodynamic energy, it can be difficult to re-establish soft nature-based coastal defences as the feedback mechanisms that protect recruits are not present[66]. Therefore, lower-energy environments, or shorelines with remnant habitat patches that can be rehabilitated are often most suitable for soft approaches (Figure 1.9).

Hybrid approaches can be applied in a wider variety of environments by taking advantage of the combination of dynamic (i.e., the ‘green’ structure) and static (i.e., the ‘grey’ structure) components that work together (Figure 1.9). For example, hard structures can support the establishment of habitats in the short-term by engineering the environmental conditions required for a particular species, which then delivers the long-term risk reduction[67, 68]. This includes providing hard substrate for the recruitment of reef-forming species, such as shellfish and corals. Hybrid approaches using dunes (such as those with a rock core) may be a preferred approach where the frequency of storms is high, or where there is limited space[64]. It is important that the hard structures are used to facilitate the habitat for long-term risk reduction, rather than using the hard component to provide the coastal defence akin to a traditional structure[69].

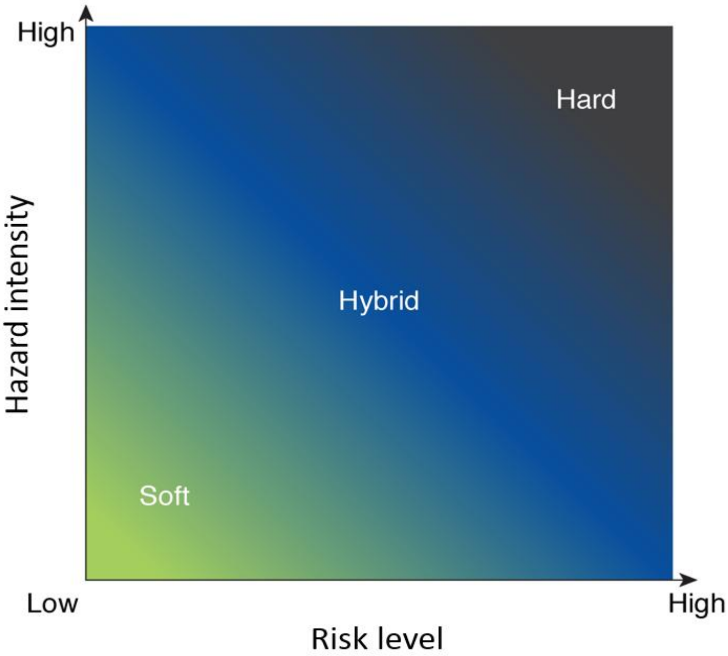


Figure 1.9: A framework for integrating nature-based methods into coastal hazard risk reduction (adapted from Morris et al. 2020[65]).

Risk Level

Risk is generally assessed by examining the likelihood of a hazard and the consequence of the hazard. Hazards pose the greatest risk where people are living on the coast. In areas that are sparsely or un- populated, an increase in coastal hazards may not cause risk to built assets. Although, a reason for risk reduction may also be to protect sites of environmental (e.g., erosion of important habitat) or social (e.g., erosion of aboriginal sites) value. Where there is a larger buffer of land between the shoreline and infrastructure, the risk level is lower, and these areas are likely to be more suitable for nature-based methods. This is because nature-based methods take time to establish, for example a soft approach using mangroves can take 5-10 years to grow, and even longer to reach full maturity[70]. Nature-based methods also require space within the intertidal to shallow subtidal to be established, and a terrestrial buffer provides adaptation space into the future. Where there is little land between the shoreline and infrastructure, increased hazards pose a high risk, and it may be too late to use a nature-based method (Figure 1.9), although consideration could be given to options such as relocation of existing infrastructure over a transition period combined with nature-based protection. Nature-based methods require forward-thinking risk reduction strategies. Hard structures can be the result of coastal management decisions only being made when infrastructure becomes high risk (a ‘fix-on-failure’ approach). This approach can be the result of the uncertainty related to decisions made now for the future. However, a wait-and-see approach can then result in the costly implementation of hard structures that may not have been required if other options were considered earlier. This is where an adaptation pathway can be a useful tool.

Adaptation Pathway Approach

Coastal urbanisation decisions have consequences over decades, and the investment in infrastructure is vulnerable to changes in climate. Given the timeframe over which these decisions need to be made, an "adaptation pathway" is an approach that can deal with the uncertainties of future conditions. These uncertainties arise from the future projections of climate change impacts and population increases, changes in societal values, and the range of adaptation options that will be available in the future[71].

Adaptation pathways are a sequence of adaptation actions or strategies that happen over time, and have the following characteristics[71]:

1. A decision point is triggered by an environmental or social change.
2. Each decision point has a series of adaptation options associated with it.
3. Once the decision point is triggered a selection for the appropriate action is made.
4. This selection then leads to the next step of the pathway, until the next decision point is triggered.
5. If appropriate, the options that are not selected at a particular decision point are still available at the decision point. This allows for flexibility and iterative decisions.

The process of developing an adaptation pathway can be based on modelling a number of scenarios for engineering solutions, which can require relatively large investment and technical expertise. There are a number of case studies, however, that focus on a low-cost community participation approach[72, 73] (Figure 1.10). The latter may be appropriate for smaller towns and local communities, while a modelling approach can be applied to large projects with significant capacity and assets to protect[72]. The adaptation pathway usually begins with low regret actions that are relatively low cost and can provide win-win situations by having additional benefits beyond adaptation. For example, the co-benefits provided by nature-based methods (Section 1: Benefits of nature-based coastal defences). While there are a number of different ways to approach an adaptation pathway, Box 1 presents the key steps that may be taken.

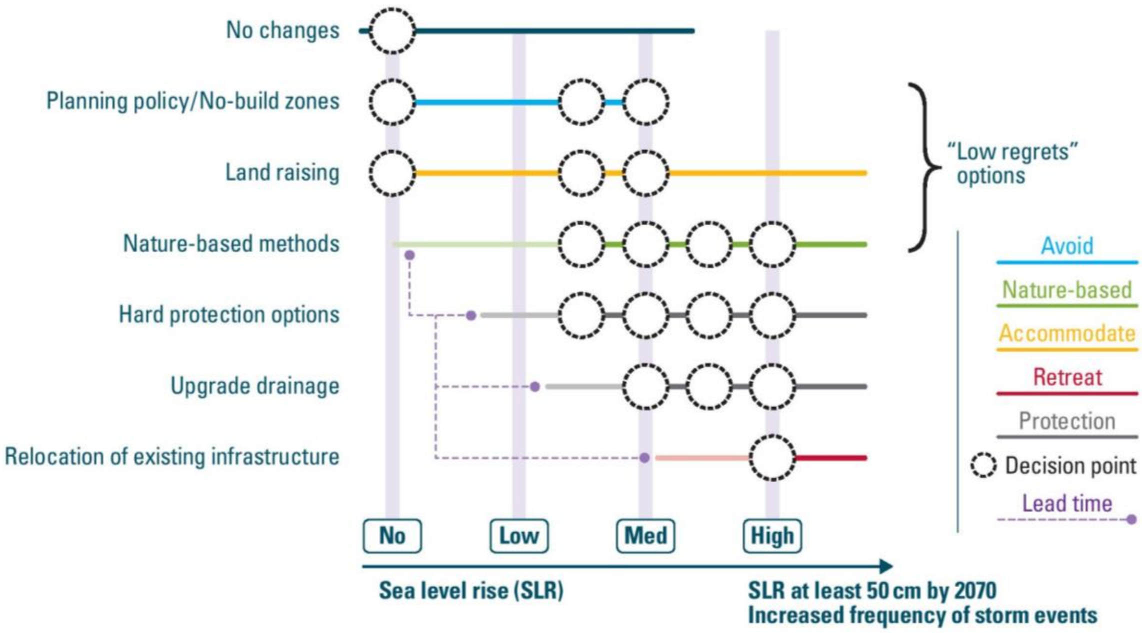
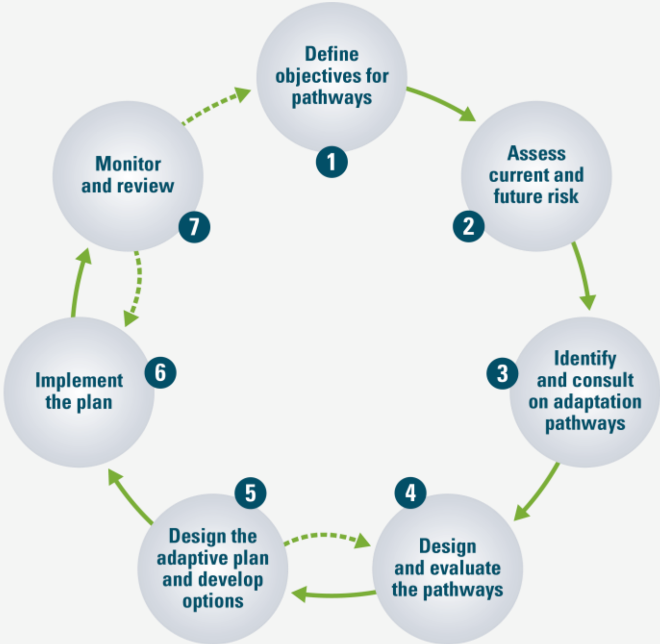


Figure 1.10: An adaptation pathway for the management of coastal development, modified from the regional climate change adaptation plan for the Eyre Peninsula, South Australia (Siebentritt et al. 2014[73]).

Box 1. Steps for a pathways approach to adaptation [71, 74, 75]



Define objectives for pathways

Define the system and its boundaries, identify stakeholders and elicit their values, determine what is to be achieved, identify uncertainties or disagreements.

Assess current and future risk

Estimate current and future hazards based on available scenarios and projections and socio-economic data, identify the thresholds and trigger points for decisions in consultation with stakeholders.

Identify and consult on adaptation pathways

Define the system and its boundaries, identify stakeholders and elicit their values, determine what is to be achieved, identify uncertainties or disagreements.

Design and evaluate the pathways

Explore the pathways and generate a pathways map, evaluate the pathways e.g., through cost-benefit analysis to guide future decisions.

Design the adaptive plan and develop options

Select the preferred pathways, identify short-term and long-term actions, build in options for iterative decisions, design a monitoring plan.

Implement the plan

Implement the short-term options.

Monitor and review

Monitor for changes in risk and available actions, implement actions as decision points are reached, reassess the plan if required.

2 Ecological and Engineering Considerations for Nature-Based Defences

The Australian coastline is approximately 35,500 km long, with more than 1,000 estuaries and 10,000 beaches[76]. The area within 7 km of the coast is also home to 50% of the Australian population[77]. The distribution, structure and diversity of ecosystems is primarily determined by climatic and oceanic factors (Figure 2.1). Nearly 90% of all beaches can be classified as wave-dominated that occur in the southern half of Australia, or tide-dominated or modified in the northern half of Australia. Coastlines in the north are subject to macrotidal ranges and generally low wave energy, however, they are affected by cyclone events and large storm surges. The tropical climate, and wave energy on the northern coasts (due to reduced exposure to large oceanic pressure systems and fetches), northeast tropical coast (moderated by the Great Barrier Reef) and southeastern Queensland (due to an extensive chain of sand islands) favours diverse and expansive coastal wetlands[78]. Along the wave-dominated, microtidal southern coastlines, dunes that back approximately 85% of Australia's beaches tend to be more extensive with far greater volumes of sand due to the higher wind- and wave- energy regimes. Mangroves reach the southern-most extent of their range in Victoria, and are confined to sheltered embayments and estuaries[78]. In contrast to the reef-building corals of the north, the Great Southern Reef is dominated by macroalgae, in particular the common kelp, *Ecklonia radiata*[79]. Reef-building shellfish were once common features of estuaries from Noosa to Perth, however, these ecosystems are currently undergoing restoration due to extensive anthropogenic-driven losses[80]. This section focuses on these ecosystems that are relevant to nature-based methods, and the parameters that are important for achieving ecological and engineering success.

Wave-dominated beach and dune system © Teresa Konlechner

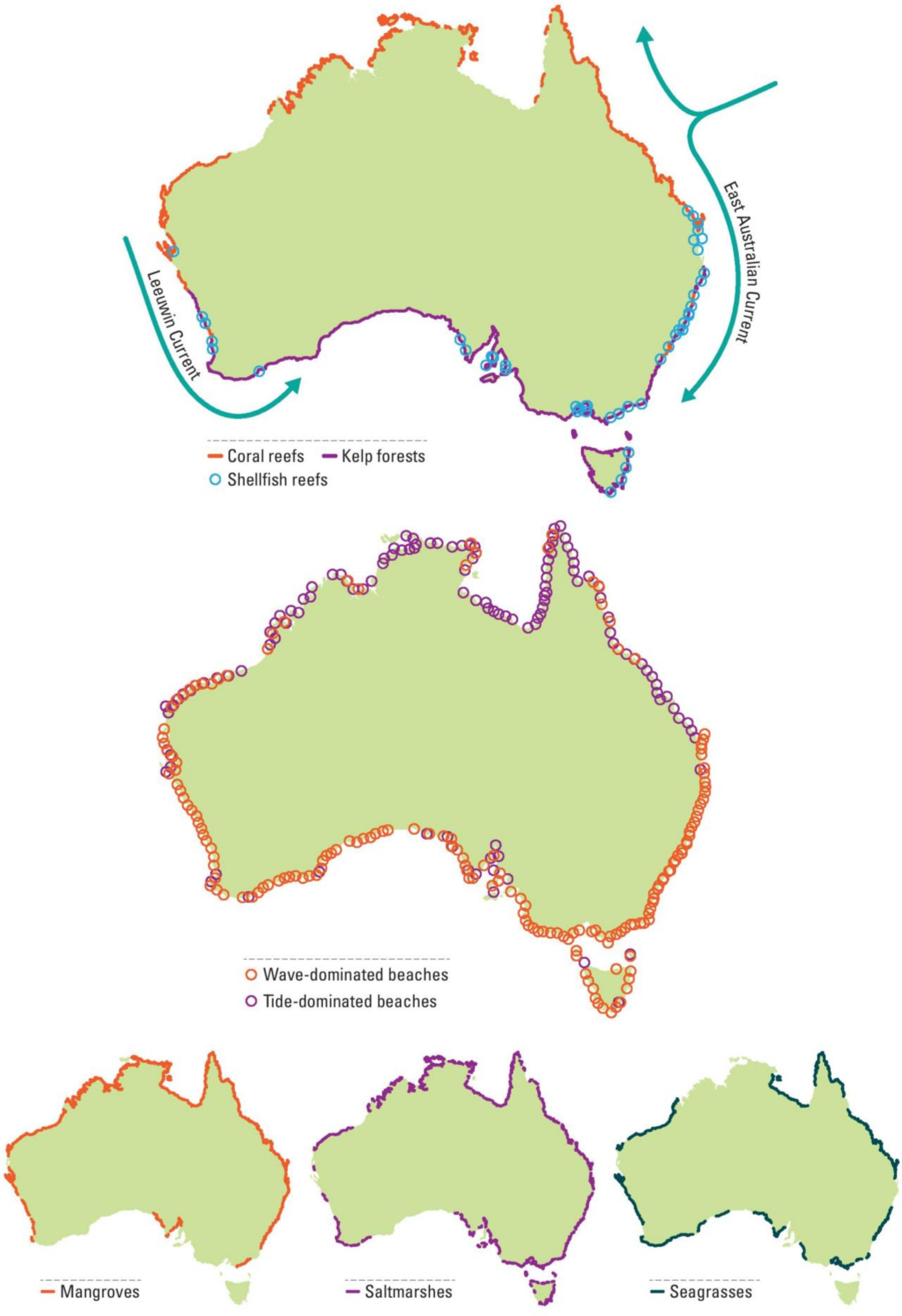


Figure 2.1 Distribution of coastal habitats in Australia.

Beaches

A beach is an accumulation of unconsolidated sediment (sand, gravel, cobbles, and boulders) extending from beneath the waterline to some physiographic change such as a sea cliff or dune field or to the point where permanent vegetation is established. In Australia, beaches are generally perceived to consist of sand, with the sand being predominantly quartz/silica in north-east and south-east Australia (including Tasmania), and predominantly carbonate in much of southern, western and northern Australia. The coast of Australia and Tasmania is approximately 29,900 km long, with 49% consisting of 10,685 predominantly sandy beach systems, with an average length of 1.37 km[81].

In Australia, the beaches can be classified based on the predominant forces that shape them[81]: wave dominated; tide modified; or tide dominated. Wave dominated beaches predominate in Australia on the open coast from about Hervey Bay, QLD clockwise to Northwest Cape (Exmouth), WA. Tide modified and tide dominated beaches prevail on the open coast of northern Australia and inside estuaries where the wave energy is low (Figure 2.1). Wave dominated beaches subject to variable wave climates and occasional storm waves and background ocean swell tend to be the most dynamic, with storm waves eroding the sand and the background ocean swell assisting with accretion. The size of the Australian coastline, combined with the variation in wave, tide, and sedimentary systems has resulted in the formation of 15 beach types within these three broader categories (see Short, 2006[81]). Macrofaunal communities that inhabit beaches are directly related to beach type[82].

Beaches in combination with coastal dunes provide protection from wave runup and flooding of, or damage to, backshore assets, and beach nourishment is common practice in Australia coastal management[83]. However, this is more often applied to maintain amenity when the landward beach limit has been armoured by a seawall or revetment. Structural nourishment without a landward hard defence is less commonly applied in Australia because there is a risk that nourishment is not feasible at the time needed (i.e. after major storm erosion events; due to logistics such as approvals, equipment or sand availability) and the nourishment volumes required may be greater for structural protection than recreational amenity (see Box 2.1). However, beach nourishment may be used in dune management programs (see section: Coastal Dunes).

Gold Coast beach nourishment © City of Gold Coast.

Habitat requirements for beaches

Beach processes are dominated by the sediment, wave and tidal regime. It is technically feasible to create or expand a beach in any regime, however, the prevailing forces will influence the shape and character of the beach. Fine silt or mud may settle onto natural or artificial beaches where low wave energy prevails, confounding attempts to create “clean” sandy beaches. On most Australian beaches in inhabited areas, the landward limit of useable beach is demarcated by a seawall, cliff or vegetated dune. Vegetated dunes are often demarcated with a fence. Despite sometimes fixed property and infrastructure boundaries, beaches are not static, and have an inherent zone of beach fluctuation and a long-term trend. These need design consideration in both traditional and nature-based coastal management.

Sediment budget

Sand is inherently mobile under the action of waves, wind and currents. The interaction of sediment fluctuation and movement with sand supply from the surrounding environment and antecedent geology is termed the sediment budget. For a beach to have originally formed requires there to have been an excess sediment budget in the past, and geological features (headlands or reefs) to trap/retain it. For the beach to remain requires the present sediment budget to be neutral or accreting, through either natural or artificial means. The main sources (and sinks) of a sediment budget are cross shore erosion and accretion, alongshore sand movement due to waves reaching the shore at an angle, wind-blown sand, onshore supply from deeper water, shell production and abrasion, rivers and estuaries, rips, headland bypassing and sometimes ocean currents.

As well as beach presence, these factors along with sediment characteristics also determine beach shape. The various zones on a beach are shown in Figure 2.2, with each zone reflecting a dynamic balance between the forces of tides, waves and wind, interacting with the geological heritage (sand and headlands), and local or imported ecosystems. Beach shape is usually defined in terms of the cross-shore profile and the planform. The cross-shore gradient below the typical wave runup limit is usually a function of the sand grain size, the tidal range and prevailing wave conditions. Coarser sand grain size creates steeper beaches. The cross-shore gradient above the typical wave runup limit is predominantly a function of the sand grain size, wind and vegetation.

The beach planform is usually a function of the wave climate (nearshore direction), headlands or end control structures (e.g. groynes) and the sediment budget. On coasts without a substantial littoral drift infeed, the planform of beaches is aligned with the predominant wave crests – “swash aligned”. On coasts with a substantial infeed of littoral drift, the planform of beaches may be at an angle to the predominant wave crests – “drift aligned”. There are many examples of beaches that change alignment in response to variable waves and/or seasonal changes. On coasts with rocky headlands and limited sand supply, short pocket beaches prevail between these headlands. Conversely, on coasts with substantial sediment and limited rocky headlands, long beaches prevail, sometimes extending for tens of kilometres.

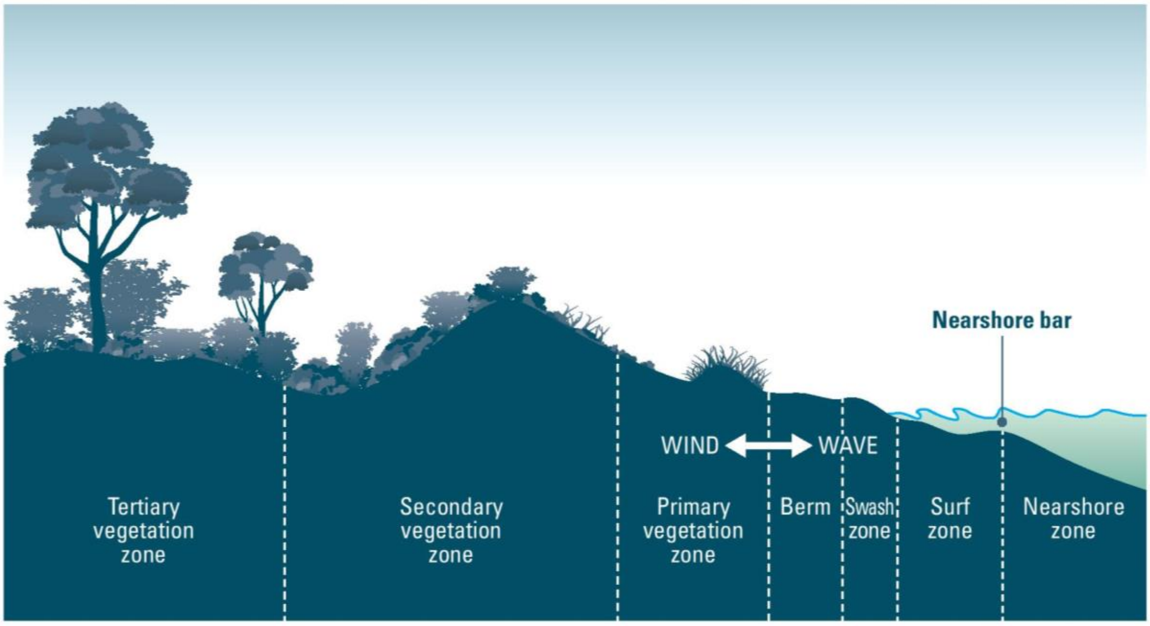


Figure 2.2: Beach profile zones (redrawn from NSW Government, 2001[84])

Accommodation Space

Beach management occurs within an active beach zone, which extends from the coastal dunes to well under the water, out to what is known as the “closure depth”. The depth of closure can be theoretically derived, and beyond which sediment transport due to waves is very small or non-existent, depending on the wave height and period. The subaerial portion of the beach (above low tide) and the frontal dunes are the primary store of sand for beach adjustment during storms. Accommodation space is more problematic when hard structures and fixed infrastructure occupies the active beach zone, which due to the natural beach dynamics can place such infrastructure at risk from erosion during sand movement.

Approaches for Hazard Risk Reduction

The extent of protection provided by beaches (and dunes) to backshore assets from short term erosion is typically estimated through an assessment of “storm demand”. This refers to the volume of sand likely to be removed from the sub-aerial beach (and dunes) (i.e. above 0 m AHD/MSL) in response to an individual storm or series of closely spaced storms[85]. Storm demand may vary both between and along beaches depending on exposure to local wave climate and the likelihood of rips etc. as well as the characteristics of the dune sediments[86]. Additional allowances are needed for other factors like beach rotation as well as recession due to the sediment budget and sea level rise over the planning period. Approaches to the assessment of these factors are evolving with probabilistic approaches increasingly used to enable communication of the full range of uncertainty, which is necessary to underpin risk management[86]. Protection from inundation and wave runup is normally defined through the dune elevation[87] although distance from the coast can also play a role in limiting the height of wave runup through friction. The primary management of beaches for protection of backshore assets or for public amenity is through nourishment.

Artificial nourishment or Replenishment

Beach nourishment is the emplacement of sand (or coarser material) onto beaches from external ‘borrow’ sites (previously defined as artificial nourishment[83]), the same sediment compartment (‘replenishment’[83]) or from scraping along the beach (see section below on beach scraping). One aim of nourishment can be to increase protection for backshore assets from erosion, recession, inundation or wave runup through creating a wider beach between the land and sea. Another aim can be to improve community amenity. Beach nourishment is considered to be a “soft” management/engineering option and usually mimics natural beach and dune systems. When compatible sand (defined below) is available for beach nourishment projects, if they are well designed, constructed and maintained, the artificial nature of the project may be undetectable to most of the community.

Sand source

Sources of sand may include the following: areas requiring dredging or excavation, such as navigation channels, lake or lagoon mouths, port expansions or basements of large buildings; terrestrial, river and estuarine sand deposits, including commercial quarries; the intertidal area (see beach scraping, below); the active littoral zone for bypassing of stable or accreting littoral features, such as natural headlands, breakwaters, training walls or groynes (see sand bypass systems, below); non-relict (active) offshore sand sources where the impacts are deemed to be acceptable; relict offshore sand deposits beyond the active littoral system, which may be the only viable option for large scale nourishment.

There are multiple methods that may be involved in extracting, transporting and placing sand (e.g., scraping with dozers or excavators; bypassing and backpassing plants; conventional excavation; trucking; dredging; reshaping with dozers; Table 2.1). The choice is usually a project-specific optimisation exercise, noting that low-impact (e.g. beach scraping) or low investment (e.g. trucking) methods are often used for initial trial projects, prior to upgrading to more complex, higher investment methods, if monitoring of initial trials deems this to be feasible. Dredging is generally used for nourishment from offshore sources. Project-specific optimisation, including frequency of renourishment, will also need to consider the need for international dredgers with high mobilisation costs.

Sand type

The best sand for a natural beach with low environmental impact will have similar characteristics to native sand in the area and is ideally surplus to the extraction area. The compatibility of borrow sand with the native sand of a beach should be considered predominantly in terms of grain size (and colour for aesthetics), but more detailed assessments can also consider shape, fall velocity, grading curve and proportion of fine material, mineralogy and biogenic fraction. While the median grain size (D50) is usually the primary consideration, the grading curve and proportion of fine material are also important to more accurately estimate project performance, as these can influence the hardness (sand compactability) of a beach and the propensity for turbid water to occur.

Sand grain size is a major factor in the cross-shore beach slope and shape, which on most wave-dominated beaches is predominantly concave-up below the water. This then has implications on sand compatibility and the required sand volumes. Further, sand grain size has an effect on dune establishment at the back of the beach (see Section: Coastal Dunes).

Table 2.3. A summary of the interventions for risk reduction using beaches. × = no; ✓ = yes; ? = information not known. The costs given are minimum – maximum.

		BEACH SCRAPING	MINOR DREDGING	SAND BYPASS PLANTS	OFFSHORE DREDGING	TERRESTRIAL SAND SUPPLY
LIMITATIONS TO ESTABLISHMENT	**Overcomes substrate limitation**	✓	✓	✓	✓	✓
	Overcomes space limitation	✓	✓	✓	✓	✓
	Overcomes time limitation	✓	✓	✓	✓	✓
	Effective against bottom-up drivers*	✓	✓	✓	✓	✓
APPLICATION	**Scalable to large areas (> 1 km^2)**	×	×	✓	✓	×
	High technical expertise required	✓	✓	✓	✓	✓
	History of use (for restoration)	✓	✓	✓	✓	✓
	History of use (for risk reduction)	✓	✓	✓	✓	✓
	Time to effectiveness (yrs)	< 1	< 1	< 1	< 1	< 1
	Unit cost (AUD m^{-3} and set up cost (AUD)	5†	5†	3†, $1 – 20M for establishment	15†, $100k – 5M for vessel mobilisation	50† (for cartage up to 10 km)
	Maintenance required	✓	✓	✓	✓	✓

*bottom-up drivers include sedimentation rates, establishment limitations; †costs per cubic metre of sand.

Sand volume

The required sand volume for beach nourishment needs consideration of the following factors: storm erosion; the sediment budget, including ongoing underlying recession, littoral drift and headland bypassing; beach rotation; rip scouring; future recession due to sea level rise; wave runup; actual composition of borrowed sand and its loss rate when emplaced; borrow area volumes available; availability of suitable plant for renourishment. If finer borrow sand is placed on a beach, then the equilibrium profile will be flatter than the natural profile, and significantly more borrow sand is required to meet a beach widening target (compared with the requirements for nourishment with matching borrow and native sand). While most beach profiles are continually changing, the long term average for a given beach, and any changes to this, may affect the biodiversity, the beach type, surf safety, surfing conditions and wave runup, and thus the design of the beach nourishment will have implications for the co-benefits that can be achieved (see Section 1.3: Co-benefits).

Mega-nourishment projects (Box 2.1) have been undertaken internationally but have not yet been undertaken in Australia. In Australia, seawalls have been more commonly used for protecting high value at risk assets, and nourishments are added to these projects to enhance or maintain beach amenity. In mega-nourishments creating a functioning beach and dune system to provide flood control is the primary objective. A trial of this approach in the Netherlands has shown that one large nourishment as opposed to frequent smaller ones can reduce environmental impact and provide enhanced beach amenity through increased width. Although, effective public communication about a change in the appearance of a beach will be required alongside these projects.

Box 2.1 Case study: Sand motor mega-nourishment, Netherlands[88]

Traditional beach nourishment involves regular nourishment frequencies of small-medium volumes of sand. This results in a frequent disturbance of the ecosystem. In a novel approach to reduce this environmental impact, in 2011 21.5 million cubic metres of sand was extracted 10 km offshore and deposited to form a 128 ha hook-shaped peninsula off the Delfland coast, South Holland. This mega-nourishment known as the 'Sand-motor' was designed to address flooding of the adjacent land through maintaining a wide beach and dune system, while preserving the ecosystem, providing recreational amenities and knowledge acquisition on these types of nature-based methods. Natural processes, such as from the waves, wind and tide redistribute this sand along the coast. The volume of sand was equivalent to the total that would have been used in more frequent nourishments over 20 years, which is the expected lifespan of the project. Current observations of sand movement suggest, however, that this could be even longer. The first evaluation in 2016, five years following nourishment showed positive results in terms of shoreline accretion, with the final assessment to be completed in 2021.

Aerial view of sand motor, July 2016
© Rijkswaterstaat/Joop van Houdt

Sand placement

The placement of sand is somewhat dependent on the extraction method and can include the following positions or configurations (Figure 2.3): dune zone; visible beach; swash and wave zone; full profile nourishment (distributed nearshore); offshore berm bar; subaqueous. Dune area, visible beach and full profile nourishment are best for reducing the erosion, inundation and wave runup hazards to backshore assets. Underwater placement is usually cheaper, with natural forces eventually redistributing the sand into a wider recreational beach, albeit without high dunes to resist wave runup.

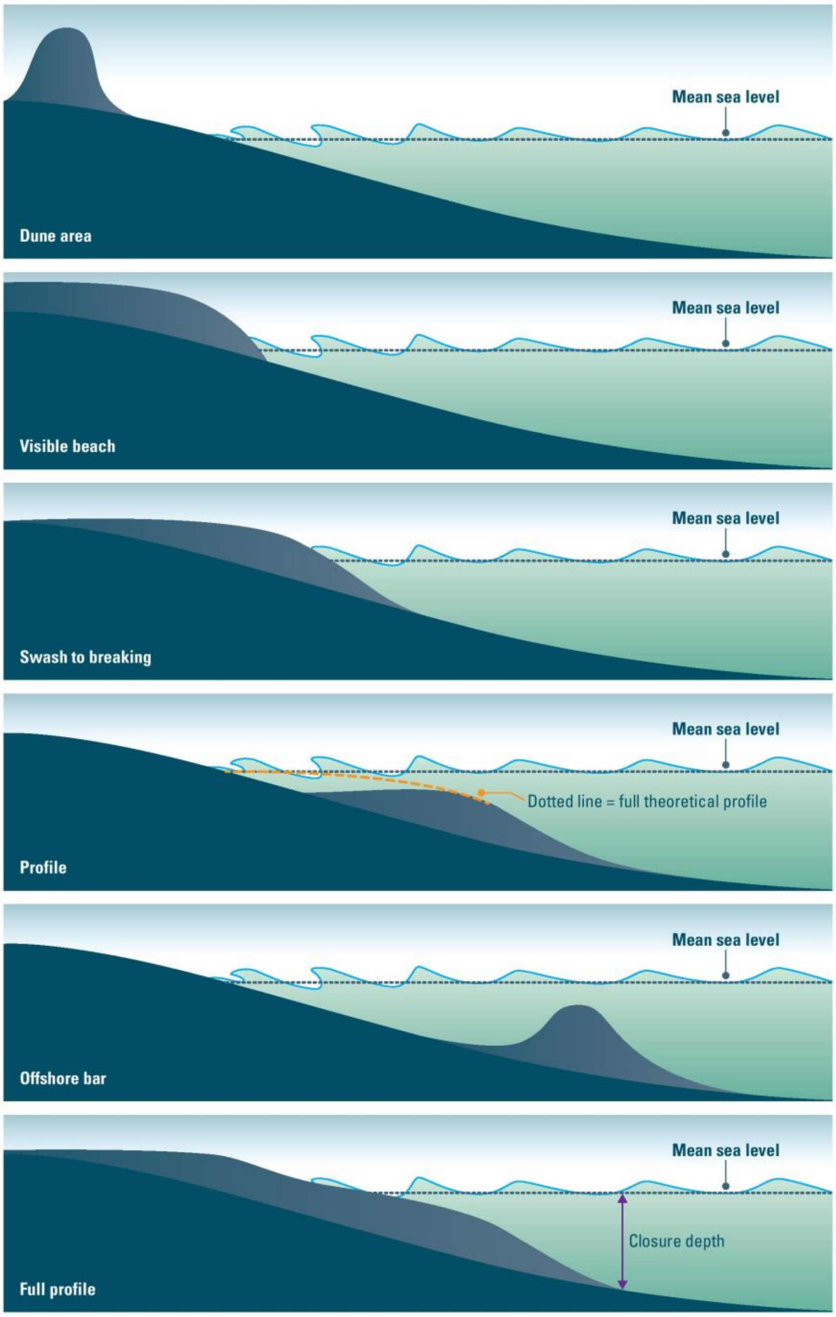


Figure 2.3: Alternative beach fill placement locations (redrawn from Smith and Jackson, 1990[89]).

Maintenance

Typical renourishment frequencies on most open coasts are 5 to 10 years, although a survey of local government authorities indicated that nourishment projects were frequently done annually[83]. Where high mobilisation costs prevail (e.g. a large international dredger), it may be prudent to place additional volume beyond the minimum requirement. Conversely, many sand bypass plants operate semi-continuously in response to prevailing wave conditions. There are also many examples of nourished beaches in reasonably closed littoral compartments (that is, bounded by headlands or groynes, and having low littoral drift) that have persisted for decades with minimal maintenance beyond dune management. The likely maintenance requirements can best be predicted by a comprehensive coastal processes study during the project's design phase, with adaptive adjustments developed during monitoring.

Beach Scraping

Beach scraping (also referred to in the literature as skimming, beach panning, nature assisted beach enhancement, assisted beach recovery and beach recycling and re-profiling) is the mechanical movement of sand from the intertidal zone to the dune or upper beach, without alteration of total volume[90]. Beach scraping is designed to mimic natural recovery processes, but decrease the time taken for recovery in comparison to natural processes. Thus, it is often done to protect infrastructure or recover pedestrian access to the beach as an immediate response to extreme weather events[83]. Beach scraping is also commonly used in combination with planting to form dunes and is valuable for raising locally low points in frontal dunes (See Section: Dunes). For beaches experiencing high recession, beach scraping is less suitable as a long-term strategy. It may be useful as a low cost and low impact strategy that can be undertaken at short notice while longer term options are investigated. The main considerations for design of beach scraping include underlying coastal processes and hazards, such as littoral drift, storm erosion, recession and wave runup. Design investigations are required to determine sustainable volumes for scraping, frequency of scraping and a target dune profile.

Hybrid approaches

Sand bypass and backpass systems

Soft and hard engineering are combined in the more than 10 sand bypass/backpass systems operating in Australia. Sand bypassing refers to mechanically transporting sand in the direction of net littoral transport. Most sand bypassing systems are associated with trained river entrances and ports, whereby sand is transferred from the updrift side to the downdrift side. The main benefits of sand bypassing are to restore littoral drift sand supply to the downdrift beaches, and to reduce sand ingress into navigation channels, thereby improving maritime safety. Examples of sand bypassing systems include the Tweed River at the southern end of the Gold Coast and the Nerang River at the northern end, and Dawesville WA. Sand backpassing refers to mechanically transporting sand in the opposite direction to the net littoral drift, which is sometimes also referred to as sand recycling. Examples of sand backpassing systems include Noosa, the Adelaide Living Beaches project, and Narrabeen NSW.

Both bypass and backpass systems require there to be an area of surplus sand e.g. updrift from a breakwater, groyne or headland, or an area of natural infill such as a river or lake mouth, and an area where additional sand is needed or can be tolerated. Both sand bypass and backpass systems require substantial engineering design and investigations, often extending over many years. Proprietary systems or system components are available, but substantial site customisation is always required. Initial trials involving dredgers or excavators combined with trucks can assist with proof of concept and optimising the design.

Performance factors for hazard risk reduction

The capacity of beaches to provide coastal protection during a storm is primarily dependent on its size and morphology (e.g., beach width, sediment volume) as well as sand properties (e.g., grain size) relative to the level of impact due to storm surge and waves. They are most effective when backed by a dune system that can act as a sediment buffer (see Section: Coastal Dunes). During extreme storm events, sediments can then be redistributed from dunes towards the beach and eventually in the surf zone forming sand bars that enhance breaking of subsequent waves, thereby reducing the impact on the coast. Generally, one of the main objectives of a nourishment when used for coastal defence is to return the coastal profile into a state of dynamic equilibrium as found on most natural beaches, where the beach (and dune) can erode during storm conditions and recover during calm conditions leading to no net long-term change. Besides these generic considerations, the design of a beach nourishment is highly site-specific and requires detailed study, for instance using numerical models.

Coastal Dunes

Coastal sand dunes are deposits of wind-blown (aeolian) sand found at the back of sandy beaches. They are naturally found in a range of contexts and forms and have long been constructed or managed to reduce flood risk on sandy coasts and provide a buffer to erosion. This is particularly the case in areas where urban development has been allowed close to the coast leading to a lack of accommodation space for natural dune development[91, 92]. The most intensively managed dunes are often narrow foredunes adjacent to urban development that span a few to tens of metres in width. Such dunes occupy a relatively narrow space between the high tide level and buildings or other infrastructure. In contrast, large transgressive dune-fields, comprising mobile dunes, may extend several kilometres inland. Most efforts using nature-based methods focus on promoting or constructing foredunes, the shore-parallel dune ridges formed on the back-shore by onshore sand transport and deposition within vegetation (also known as primary or frontal dunes)[93]. Sand dunes help to protect against coastal flooding by forming a physical barrier against elevated nearshore water levels, e.g., due to storm surges, and wave runup (see Section 1.2). In addition, they provide a reservoir of sand to help replenish the beach during severe erosion events[5, 94].

Vegetation is critical for forming and maintaining dunes and for increasing coastal resilience to erosion[95]. Plants intercept sand transported by wind, promoting sand accumulation and promoting further dune growth[96, 97]. Plants also stabilise the sand surface, which reduces wind erosion of the dune surface and promotes dune stability. Buried shoots, roots and other below-ground structures such as rhizomes may increase soil cohesion effectively increasing erosion resistance[98, 99], while the above ground structures (leaves, stems) attenuate wave energy during wave overtopping (i.e., when wave runup overtops the dune crest) and erosion events[100, 101]. Stems and leaves alter turbulence and flow patterns and, therefore, patterns of scouring, erosion and sediment accumulation[95]. Vegetation also promotes dune recovery after erosion by facilitating incipient dune development[102, 103].

A vegetated dune in Port Phillip Bay, Melbourne with access ways and fencing to control pedestrians.
© Teresa Konlechner

Sand dunes occur on approximately 85% of the Australia's sand beaches. They are found in all regions and across all climatic zones, including the humid northern coasts of Australia[76] (Figure 2.1). Dunes tend to be more extensive with greater volumes of sand due to the higher wind- and wave- energy regimes along the wave-dominated, microtidal southern coastlines. Dune development is less apparent along the northern coasts of Australia, however, relatively large dune fields do occur at various locations, including north of Broome, the Dampier Peninsula, the Tiwi Islands and the complex transgressive dune fields of Fraser Island and Moreton Island in Queensland. In contrast to other habitats that can be used for nature-based risk reduction (e.g., corals and mangroves), there are not strong latitudinal limits or regional patterns on foredune development in Australia. Controls on dune formation or morphology at any given location in Australia reflect local scale variations in sediment supply, wind and wave climate, underlying geology, beach morphology and vegetation cover rather than broad scale regional factors[76].

Habitat requirements for coastal dunes

In general, foredunes can form on any sandy coast provided there is enough space above the average reach of waves (accommodation space), winds are strong enough for sediment mobilisation (sand supply), and where primary colonising vegetation is present to trap and hold some or all of that sand (vegetation)[93]. These factors are interrelated and interact to control the rate of dune growth, dune size and dune shape. All other factors being equal, space and time are important to allow dunes to increase in size to create foredunes large enough to survive storms of greater frequency/magnitude[104]. Higher rates of sand transport results in a potentially faster progradation rate of dunes, while the type of vegetation (growth form, density and cover) determines the vertical rate of dune growth and the resulting dune form[93, 105]. Many of these habitat requirements can be manipulated to enhance dune development and promote the formation of desired dune forms.

Accommodation space

Accommodation space refers to the space at the back of the beach where a dune can form or be created. The width of the accommodation space determines the size and morphological complexity of the dunes. As a rule, a larger accommodation space allows for relatively high and wide dunes, and for more complex dune shapes and increased dynamism. Consequently, this leads to increased resilience to flooding and erosion, and support of more diverse plant communities[104, 105]. Accommodation space is predominantly a function of beach width, with wider and higher elevation beaches providing more space and time above the usual reach of waves for dunes to develop. Dunes do not always occupy the full accommodation space available, and dune development is typically slow relative to rates of beach recovery after erosive events. On metropolitan coasts, the recreational benefits of beaches are often prioritised over dune building restricting dunes to a narrow zone at the rear of the beach. Beach grooming (using heavy equipment to remove e.g., debris, seagrass), beach driving or trampling by pedestrians leads to compaction of the beach making it more difficult for the wind to transport sediments to the dunes, and thus limiting dune growth[106]. The proximity of infrastructure close to the water restricts accommodation space on many coasts[107]. Eroded dunes can reform, even when wave attack and periods of dune destruction are frequent. On constrained coasts, however, the intrinsic rate of aeolian sand accumulation may be too slow to allow dunes to develop before the next erosion event. In such cases, management such as beach nourishment or the mechanical construction of dunes may be essential to offset the restrictions of a narrow beach width[107].

Sediment supply

Sediment supply for dunes is a function of the sand on the beach and the transportation capacity of the local wind regime. Dunes are initiated through the entrainment and landward transport of sand from the beach by onshore winds. Beach sediment around the Australian coast is predominately fine to medium sand, with an overall mean size of 0.4 mm[81]. Nominally a wind velocity of >20cm s^{-1} is required for transport of sand of this size; although sand transport potentials, hence sand supply to the dunes, varies considerably between beaches. The rate at which sand is transported from the beach to the back of the beach to the foredune is determined by the fetch, the width of dry beach (intertidal and berm) over which the wind blows, sand size and shape, sand composition (quartz or carbonate), beach morphology (e.g., slope) and the frequency and strength of onshore winds[108, 109]. Every beach will be unique, but beaches with a high tidal range, wide open-coast beaches, finer sediments, and subject to strong oblique onshore incident winds are generally optimal for dune development. Although dunes are found in all regions of Australia these conditions are more generally met on the temperate coasts south of the tropics, and sand transport occurs more frequently and in greater volumes along the western windward and southern coasts of the continent due to increased exposure to strong onshore winds. Limitations in sediment supply can be partly overcome by increasing the width of the beach (i.e. through renourishment, although care to use sand of a size suitable for aeolian transport is critical), or by depositing sediment directly and reshaping it using earth-moving equipment[110].

Vegetation

Plant communities on foredunes usually consist of low salt- and wind-tolerant grasses, succulents and creepers, the so-called primary or pioneer dune species, which grade landward into shrub and forest covers. Beaches and foredunes are stressful environments for plants, characterised by frequent substrate disturbance by wind and waves, burial by sand, drought, and salt stress[111]. As such, they support a relatively restricted flora and in most regions of Australia only a few species play an important role in foredune development. These species vary with climatic zone and local vegetation guides should be consulted to identify appropriate species for local foredune conditions. Excessive burial, drought stress, inundation by seawater, mechanical damage by waves, and trampling are key restrictions of vegetation growth in beach foredune environments, with juveniles particularly vulnerable to high levels of these environmental stresses[111]. Only in the most extreme cases, however, are environmental conditions completely limiting for vegetation growth on Australian dunes with foredunes generally sustaining a relatively robust vegetation cover[76].

Approaches to hazard risk reduction

Sand dunes protect landward communities, habitats or other assets from erosion or flooding by forming a physical barrier against elevated nearshore water levels, e.g., due to storm surge and/or high energy sea-swell waves. To be effective, dunes must be capable of withstanding periodic storm damage to both vegetation and morphology due to erosion. Increases in dune size relative to the frequency and magnitude of storm impact enhance the resilience of dunes to erosion. Hence, approaches to dune-based hazard risk reduction mostly focus on strategies to protect or enhance existing dunes, or to promote aeolian sand deposition and dune formation. While most protection services of dunes are provided by the landform (i.e. the mound of sand at the back of the beach), protecting or re-establishing a vegetation cover is an integral component of most coastal defence projects because of the role vegetation plays in facilitating sand accumulation, dune growth, and surface stability. In recent years, recognition that vegetation also plays a direct role in reducing rates of dune erosion and overwash (i.e., landward flux of sediment over dune top in case wave runup exceeds dune crest) during storm events has increased[95], further emphasising the importance of incorporating vegetation in dune projects.

The choice of appropriate approach depends on several factors including the habitat suitability of the coast for dune building, the presence or absence of existing dunes, and the space and time available for dunes to increase in size relative to the frequency of hazards. Sometimes, practical considerations play a substantial role in decision making. For example, the desire for residents with foreshore property to have unobstructed views to the ocean is one of the most limiting factors for dune development in areas with high coastal development. Approaches for dune creation or restoration along urbanised coastlines typically rely less on natural aeolian processes and more on human interventions such as sand fences, vegetation planting or building dunes with earth moving equipment to promote dune growth as the space and time for natural dune-building processes decreases (Table 2.2).

Rehabilitation

Dune rehabilitation refers to the restoration of dunes, from a degraded to a less degraded or unimpaired state, in order to gain the greatest coastal protection benefits[112]. This can be done by removing anthropogenic stressors to promote the recovery of damaged dune vegetation, to repair damage to the form of the dune, and to re-establish natural dune processes of aeolian sand transport from the beach to the dune and recovery following erosion. Vehicle and pedestrian traffic are a common cause of vegetation loss[113, 114] and promote the development of topographically low points in a dune form, which can act as conduits for overwash[115]. Restricting vehicles and managing pedestrian traffic through the use of elevated walkways, fences, signage and designated parking areas are effective methods of managing these pressures[116, 117]. Weed control, where non-native species have replaced native dune building species, can promote the formation of more protective dune morphologies and allow for improved aeolian processes[118, 119]. Reshaping the foredune to a more protective shape, for example to increase height and decrease the steepness of the seaward face, or to improve the natural functioning of the dunes may be required at highly degraded sites[84, 119]. Remnant native plants can usually regenerate once stressors are removed but replanting will be required on highly disturbed sites or where regeneration is too slow[84]. A single one-off rehabilitation intervention will seldom be sufficient. Maintenance of fencing and pedestrian access, sustained weed control, evaluation of the efficacy of rehabilitation actions, and monitoring for new threats are important for long-term rehabilitation success. Complete rehabilitation of very degraded sites may take several years[117]; however even degraded dune systems provide some protection against coastal hazards. Full rehabilitation to a "pristine dune state" is not always needed to improve the protection capacity of a dune-system although it may be desirable for promoting other co-benefits.

Table 2.2. A summary of the interventions for risk reduction using coastal dunes. × = no; ✓ = yes; ? = information not known. The costs given are minimum – maximum.

		RESTRICTING VEHICLES, MANAGING PEDESTRIANS, CONTROLLING GRAZERS, WEED CONTROL	REVEGETATION	SAND FENCES	MECHANICAL CONSTRUCTION OR RESHAPING	DUNE CONSTRUCTION WITH HARD CORES
LIMITATIONS TO ESTABLISHMENT	**Overcomes substrate limitation**	×	×	×	✓	✓
	Overcomes propagule limitation	×	✓	✓	×	×
	Overcomes space limitation	×	×	×	×	✓
	Overcomes time limitation	×	×	×	✓	✓
	Effective against top-down drivers*	✓	×	×	×	×
	Effective against bottom-up drivers*	×	✓	✓	✓	✓
APPLICATION	**Scalable to large areas (> 1 km^2)**	✓	✓	✓	✓	✓
	High technical expertise required	×	×	×	×	×
	History of use (for restoration)	✓	✓	✓	✓	✓
	History of use (for risk reduction)	✓	✓	✓	✓	✓
	Time to effectiveness (yrs)	?	3-10	1-5	Immediate	Immediate
	Set up cost (AUD m^{-2})	?	30-60	10-60	40 - >?	? - >1000
	Maintenance required	✓	✓	✓	✓	✓

*top-down drivers include grazing, trampling; bottom-up drivers include sedimentation rates, establishment limitations

Using vegetation to promote dune building

Planting vegetation to promote sand accumulation, to enhance resistance to erosion, to stabilise the sand surface, or to reform a dune will be required where the foredune vegetation is degraded or absent or where the natural processes of vegetation establishment are restricted. Species selection depends on the primary purpose of vegetation; with different traits important for sand accumulation, stability, hydrodynamic or storm recovery purposes (Table 2.3). The species that are most useful in building foredunes are typically graminoids that form a uniform canopy and respond positively to burial[111]; although low creepers and woody plants increase in importance in tropical regions[120]. Plant species type is important in determining morphological development with tall species tending to produce higher, more hummocky peaked dune forms than lower, more spreading, rhizomatous plants[93, 121]. Other considerations for suitable species selection include ease of propagation, survival rate, and compatibility with environmental characteristics of the transplant site[107]. Locally native species (e.g., *Spinifex sericeus*) should be used over exotic species (e.g., *Ammophila arenaria*). In contrast to revegetation for biodiversity purposes, the aim of hazard-risk reduction projects is to establish a uniform vegetation cover as quickly as possible. This is most readily achieved through the use of transplants rather than seed. Young nursery-grown plants grown from locally sourced seed are often used; however, the availability and cost for large-scale restoration projects can be limiting. Divisions or cuttings of established plants can provide a cost-effective source of propagules, although care is required to not degrade the source site.

Table 2.3. Considerations for the selection of plant species for coastal protection (adapted from Feagin *et al*., 2015[95]).

PRIMARY PURPOSE	CHOOSE SPECIES BASED ON THEIR ABILITY TO:
Sand accumulation	Promote sand accretion/build elevation Develop high dunes versus low hummocks Fit within a heterogeneous array of different successional stages
Soil stability	Increase organic matter and water content, reduce soil bulk density Promote mycorrhizae, increase effective grain size of non-cohesive sand particles Form dense long-lived belowground root/rhizome systems, high aboveground cover
Alter hydrodynamics	Attenuate waves and alter water velocities according to: stem height, flexibility; leaf area; above ground biomass; overall plant architecture Reinforce, abrade, or loosen soils according to: root diameter and density and configuration; belowground biomass
Storm recovery	Physiologically respond to storm erosion according to: compensatory stimulation of growth; modes of post-erosion reestablishment

Transplant survival can be low or highly variable due to variability in rainfall, wind erosion of the substrate, excessive burial by aeolian sand deposition, burial or mechanical damage by waves and overwash, drought, damage by people or by grazing animals[99, 104, 122]. Survival can be enhanced by appropriate species selection (i.e., species adapted to local climate conditions), utilising tillers with root and rhizome material attached and larger plants rather than seedlings, supplemented watering or planting during seasons when water deficits are low, managing people to avoid physical damage (i.e. from pedestrian trampling or vehicles), and protecting animals where grazing is likely to cause significant mortality (i.e., where rabbits are abundant)[104, 122]. Composting with seagrass wrack or inoculating transplants with arbuscular mycorrhizal fungi have been suggested as additional means of improving plant growth and survival on restored dunes[99]; however, their ability to improve planting success or the feasibility of applying these approaches at the required scale of most dune projects is not yet known.

Careful consideration of the spatial pattern and density of planting is necessary, as these influence local patterns of sand accumulation and dune development. Dune planting guidelines in New Jersey, USA recommend a staggered planting pattern with a uniform alongshore density and an increase in plant spacing from approximately one plant per metre at the toe of the foredune to ~ 30 cm at the foredune crest[123]. This type of plant spacing results in a wider, gently sloping dune with alongshore uniformity in form[123], however, the appropriate planting density at any given location will be site specific.

Revegetation projects lend themselves to partnerships with community groups, with volunteers participating in planting and maintenance. The man-hours provided by community members help offset costs associated with revegetation projects, as well as yielding other benefits including increased awareness and support for the need for dune protection[124].

Using sand fences

Sand fences encourage sand accumulation and promote foredune growth. The use of fences to control sand drift is attractive to managers as they are usually effective, inexpensive, easy to deploy, and their effects can be seen quickly[125]. Their effects on sand transport can persist even when the fence is buried, particularly if vegetation has established in the interim[126]. They are 'accepted' coastal structures (in contrast to seawalls and breakwaters, for example), and generally attract relatively minimal opposition from the public[126]. Limitations of sand fences include the loss of views if dunes become too high; the need for ongoing maintenance and repairs; decreased efficiency through time as sand accumulates; barriers to people and faunal movement; and the need for increasing numbers of fences as existing fences become removed from the original zone of sand transport[126]. Sand fences can be visually intrusive, particularly where damaged fences are not removed). These adverse effects, however, can be avoided in many situations through careful fence design and deployment[126].

Sand fences are constructed from many materials (e.g., wooden slats, plastic and jute mesh, and saplings and branches, such as brush), and can be deployed in a range of configurations (e.g., straight vs. zigzag, alongshore, diagonal or perpendicular to the shore, single or a series of multiple fences[126]). The efficiency of a sand fence is a function of the fence design; particularly fence porosity and height[127]. A fence design with a porosity of about 50% is generally accepted as optimal[126, 127]. The lifespan of the fence is proportional to its height since the fence will gradually lose its trapping function once sand accumulates to about 80% of the fence height[127]. Most fences are between 0.6 and 1.3 m high[128]. Fence length, width, configuration, single or multiple fence row deployment, separation distance between fence rows, and orientation relative to the wind are also important[127, 129]. The most appropriate design for any sand fence will be site-specific and depend on a range of factors including the purpose of the fence (e.g., dune building vs sheltering), the preference of the local community, and the intended life-span and cost.

Mechanical dune construction

Dunes can be constructed by directly depositing sand and reshaping it using earth-moving equipment if the natural processes of dune building are too slow, even when assisted by vegetation and sand fences, or a project must be completed in a set period. These landforms, sometimes called dune-dikes, have the advantage of being able to be created quickly in environments where new dunes could not form, or survive long enough, to provide protection against storms. They can also be designed to optimise flood and erosion protection services[104]. Ideally, earth-moving equipment will be used to create a dune ridge for initial protection, but with the goal of allowing a more natural dune to gradually evolve[107]. Dune construction should usually be combined with vegetation planting to help stabilise the constructed dune surface and promote subsequent dune growth.

The cost of dune construction will vary depending on the dimensions of the desired dune, the length of coastline to be protected, the hire of earth-moving equipment, and the availability of suitable material (sediment) for construction. Additional costs include those associated with vegetation planting and ongoing maintenance. In ideal conditions the created dune will be sufficiently large to survive subsequent erosion events, with the capacity to recover when eroded. However, in conditions of limited sediment supply and frequent erosion, repeated rebuilding to maintain the desired protection services is usually required.

Hybrid dunes

In some cases, it may not be possible for dunes to provide the desired protection services alone. Hybrid dunes which incorporate a hard core designed specifically to resist erosion have been used on coasts where there is insufficient space and time for a purely natural approach, and where erosion poses an immediate threat to high-value assets[130, 131]. Cores are often constructed using geotextile tubes filled with sand, although other materials including clay, gabion baskets and rock have also been used to form the core[132]. An example of this approach is used on the Gold Coast (termed the ‘A-line’). These structures are installed along the toe of an existing dune or scarp and covered with sand to mimic a natural foredune[132, 133]. The cores are most often placed in a horizontal configuration but can be sloped or stepped to help dissipate wave energy when uncovered and in direct interaction with the waves[134]. Vegetation is usually planted on the surface to aid in anchoring the nourished sand and to promote aeolian sand capture and natural dune growth. The goal of the hard core is to provide a “line of last defence” against erosion[133], while still allowing for a degree of natural dune functioning. Traditional hard-shore protection structures positioned landward of existing dunes or that become covered by beach accretion or aeolian transport when the local sediment budget is enhanced are also considered hybrid shore protection structures[132].

Hybrid dunes that incorporate a core are recommended for shoreline hardening projects as a low-cost alternative that can be deployed and/or removed more expediently compared to traditional engineered structures[133]. They are most suitable in situations which require a greater degree of protection than provided by sand dunes alone, where erosion occurs too frequently for dune survival, and where the use of traditional hard structures can be avoided. There is, however, relatively little information on the implications of incorporating hard cores into dunes for coastal protection[135]. Hard cores have been effective at preventing erosion during storms, but must be designed to be appropriate to the energy level of the waves where they are deployed[136, 137, 138, 139]. Some studies have shown decreased beach volumes, increased scouring, and accelerated erosion landwards of the hybrid dunes, suggesting that they act more like ‘hard’ shoreline protection methods than natural dunes during erosion or overwash events[133]. In addition, because hybrid dunes are usually deployed to prevent erosion in marginal settings, they require ongoing maintenance to maintain a protective sand and vegetation cover [130, 133, 136, 138]. Hybrid dunes have been received positively by the public because they mimic the aesthetic values of natural dunes, while being perceived to delivering enhanced protection values[138]; and so, they gain support for implementation in preference to more permanent structures. However, this positive perception remains only if they remain covered by sand[138]. A further limitation of hybrid dunes is that they have limited adaptive capacities, more similar to traditional hard structures that “hold the line”, compared to non-hybrid dunes. The cost of constructing hybrid dunes will vary depending on the choice of core material and the length of coast to be protected. They are less expensive compared to hard structures, even when maintenance costs are included in the cost estimate; but are more expensive than mechanical construction and actions to promote dune construction[133]. As such they are more suited to deployment over relatively short areas of coastline where high value assets are threatened.

Performance factors for hazard risk reduction

The impact of storms and other extreme events on coastal dunes has been the subject of considerable research. As such, we have a reasonably good understanding of those factors that enhance dune survival or reduce the impact of the hazard in different geomorphic settings. These factors can be used as relative indices that link dune condition to a greater or lesser degree of coastal protection; although the actual protective capacity of a dune will reflect a complex range of interrelated factors including the storm characteristics, sediment characteristics and pre-storm beach morphology. Storm protective capabilities of dunes have been evaluated based on their capacity to protect land situated immediately inland of them during extreme events, their resistance to erosion, and by their capacity for self-repair after an erosion event[140, 141, 142, 143]; with dune morphology, vegetation and “natural dynamism” identified as important factors.

Dune morphology

The protection capacity of dunes is positively correlated with size[5, 142, 143]; with increased height, width, volume, and alongshore uniformity in form correlated with increased levels of protection. The relative importance of these metrics depends on the nature of the hazard (i.e., wave overtopping vs. erosion) and mechanism by which dunes provide the desired protection services (i.e., act as physical barrier, a reservoir of sand to dampen wave energy when eroded, or as a roughness element to reduce wave energy; see Section 1.2). Some studies suggest that higher dunes may be more protective than lower dunes against dune overwash[142, 144], with dune height being more important than beach-foredune width for determining protection capacities[142]. However, wider dunes with a greater volume of sand are usually more strongly correlated with reduced rates of storm-related erosion[141, 143, 145]. Dune width, height, and therefore volume, are not uncorrelated. High dunes require a wide base to form[105], although factors such as the beach sediment budget and vegetation have been found to enhance vertical over horizontal dune growth in some situations[121, 146]. Alongshore uniformity in form is also important for reducing dune overwash penetration and localised erosion; however, such topographic variability is also important for increasing the longer-term adaptability of foredunes by allowing for inland sand transport and should be allowed where possible.

Vegetation cover and density

The presence of a healthy vegetation cover can be an indirect indicator of foredune protection capacity because of its association with increased dune volume and height and promotion of recovery after erosion events. A variety of measures have been suggested as increasing protection capacity, including enhancing species richness and zonation, maximizing the presence and vigour of dune-building species, or planting at optimal densities to promote sand accumulation. How these factors directly translate to hazard risk reduction, however, is seldom studied. There is a growing recognition that vegetation can also directly influence the rate and magnitude of dune erosion and overwash[95, 99]. The role of dune vegetation in attenuating wave energy and limiting overwash is not yet well understood[95, 99]; hence best practice guidelines for species selection and planting patterns to enhance this capacity of dune vegetation are yet to be developed. Laboratory tests using small scale wave-flume tanks have found vegetation on the seaward dune face decreases the total volume of eroded sand and the rate of dune scarp retreat by as much as ~30% compared to unvegetated dunes[99] by decreasing wave runup, and overtopping and subsequently limiting rates of sediment flux due to overwash[100, 147, 148]. The above ground portions of plants appear to be the primary mechanism through which erosion is reduced[101], although below-ground structures also confer erosion resistance[98, 149]; and the effect may be species specific[150]. The extent to which these experiments scale to field conditions is as yet unknown.

Allowing for "natural dynamism"

Dunes are dynamic systems, which are stressed during episodes of coastal erosion, drought, fire, and periods of exceptional windiness. This dynamism promotes landscape complexity, habitat diversity and geomorphic resilience, as well as adaptability to changing environmental conditions. Key processes that are indicators of the resilience of dunes to erosion include their capacity to recover following an erosion event, allowing for aeolian sand transport and deposition, and the landward transfer of sand either by overwash or by downwind dune migration[110]. Wherever possible managers should allow for the dynamic nature or dynamic potential of dunes. The design of a dune management program might consider the potential for dunes to migrate and establish new forms and positions. Naturally this process may be in conflict with the adjacent land use and difficult or impossible to achieve at many locations. Coastal dunes in greenfield sites should be allowed to migrate, if circumstances permit, to maintain a full range of ecosystem services. Where foredunes are confined by land use the options are significantly reduced. In the United States, Netherlands and elsewhere, including the Gold Coast, one solution has been to maintain a positive beach sand budget by nourishing beaches with sand pumped onshore or alongshore (see Section: Beaches). Natural processes are then allowed to function as long as the critical beach width and dune height are maintained.

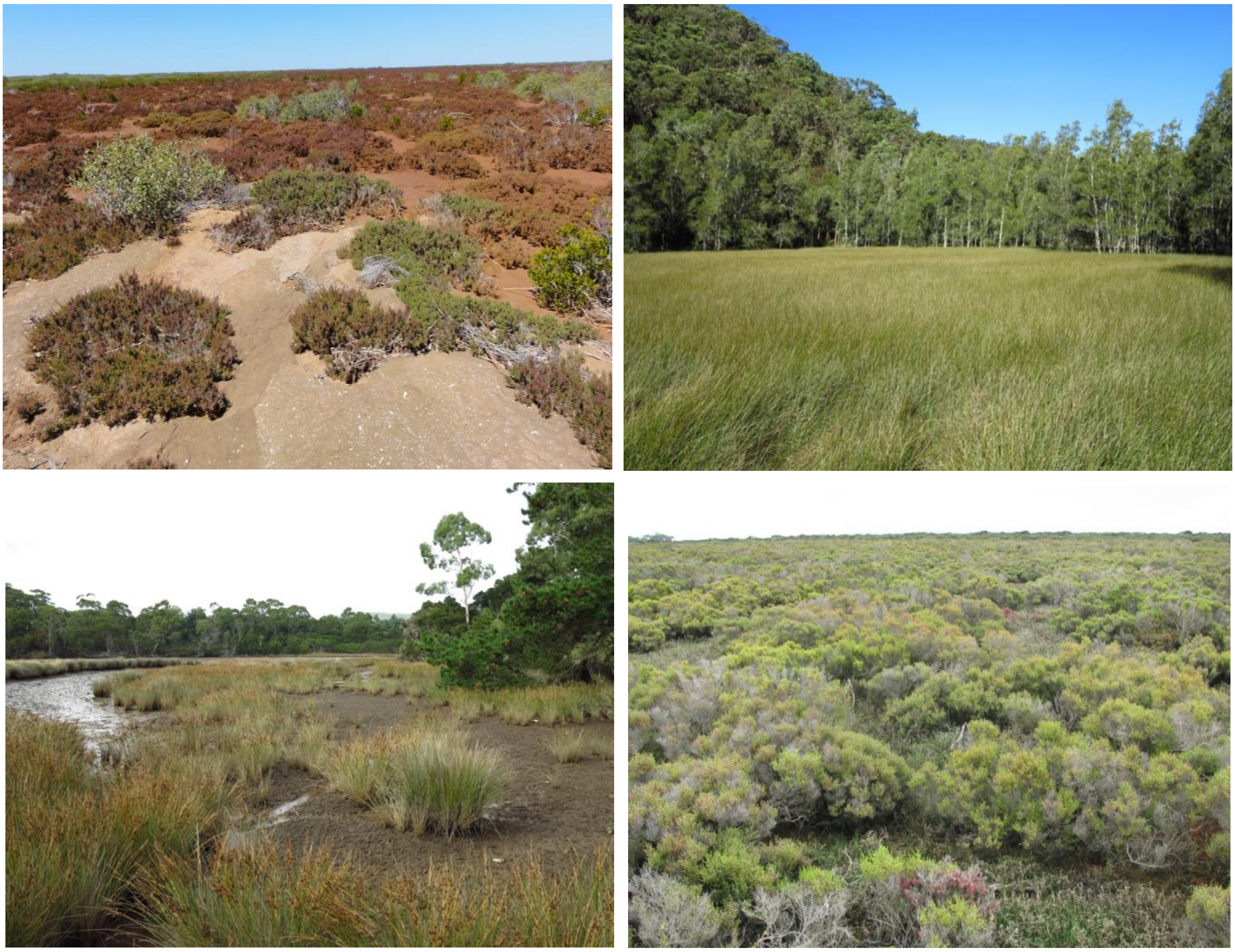
Saltmarshes

While the definition of coastal saltmarshes varies among jurisdictions in Australia, for the purpose of this guideline they are considered to have the following four characteristics: (1) found on littoral sediments on low-lying, low-energy coasts; (2) subjected to tidal or weather-affected inundation by seawater, which results in soils that are continuously waterlogged or at least temporarily moist; (3) have saline or hypersaline soil (i.e., having a salinity greater than that of seawater, or >35 g L^{-1}); and (4) are characterised by plant communities that consist completely or mostly of halophytic species (i.e., those that can grow, and often reproduce, in soils containing appreciable amounts of salt).

Coastal saltmarsh occurs around much of the Australian coastline (Figure 2.1). Unlike mangroves, coastal saltmarsh is common in Tasmania and southern Western Australia, and even along parts of the Great Australian Bight. On a nation-wide basis, coastal saltmarsh is most extensive in northern and western Australia, but large areas are found in all States and in the Northern Territory. In regions of the coast where it co-exists with mangroves, coastal saltmarsh is frequently found in wide bands or mosaics on the landward side of the mangroves, which often occur as a fringe along the shoreline.

The perennial flora of Australian coastal saltmarsh is overwhelmingly dominated by species in the plant families Amaranthaceae (i.e. chenopods, dicots) and Poaceae (i.e. grasses, monocots)[151]. Coastal marsh is, however, very structurally variable across Australia. Low-lying shrubs are common in coastal saltmarsh from northern Australia, as well as on the mainland of southern Australia and in Tasmania[152]. Chenopod-dominated forblands are exceptionally common across the country; rhizomatous rushlands are also extensive along some parts of the coast. The diverse habitat structure provided by saltmarsh has significant wave damping effects, as well as sediment accretion and stabilisation[153, 154]. In some saltmarshes, especially in Tasmania, and to a lesser extent Victoria and South Australia, the introduced (North American) saltmarsh grass *Spartina* has invaded so successfully that it is now the dominant groundcover in some estuaries.

An invasive *Spartina* dominated saltmarsh, Rubicon River, northern Tasmania. © Paul Boon



The diversity of native Australian coastal saltmarsh. From top left to right: chenopod dominated marshes with scattered mangroves of *Avicennia marina* in northern Australia © Norman Duke; *Baumea*-dominated in southern Australia fringed by Swamp She-oak *Casuarina glauca*; *Tecticornia*-dominated in southern Australia; and *Juncus*-dominated in northern Tasmania © Paul Boon

Habitat requirements for coastal saltmarshes

Temperature and rainfall

In contrast to mangroves, which have tropical origins[155], the plants that grow in coastal saltmarsh are fundamentally of a temperate origin. This is reflected in the pattern of decreasing floristic diversity of saltmarsh with decreasing latitude[156]. At a mean minimum temperature threshold of 8°C, saltmarsh diversity is greatest, which declines with increasing temperature[156].

Saltmarsh communities are also affected by environmental salinity, which in turn is influenced by both temperature and rainfall. Coastal saltmarshes in northern Australia often appear different to those from southern Australia, seeming to be more frequently hypersaline, and this is likely to be a function of higher air temperature, less equitable rainfall and greater evaporative losses in northern Australia. Rainfall patterns also influence the development of coastal saltmarsh in temperate southern Australia. In

Victoria, for example, there is a distinction between the 'dry' type, present in the central-western parts of the State, where low summer rainfall and high temperatures lead to intensely hypersaline conditions, and the 'wet' type more common in the east of the State[157]. The vegetation in the western 'dry' type is often dominated by *Tecticornia pergranulata* and *Tecticornia halocnemoides*, species able to survive intense hypersalinity, but the eastern 'wet' type is usually dominated by samphires such as *Sarcocornia* spp., *Suaeda australis* and *Tecticornia arbuscula*. This geographic differentiation of 'dry' and 'wet' forms of coastal saltmarsh in Victoria is a direct consequence of patterns in rainfall and evaporation across the State. Given the large diversity of saltmarsh in Australia, it is important to consider the appropriate species for the area when restoring these ecosystems for coastal hazard risk reduction.

Elevation and tidal regime

The relationship between the level of the land and neighbouring water body is the critical factor controlling the formation, structure and function of coastal saline wetlands[158]. These factors not only delimit the upper boundaries of coastal saltmarsh and its interface with the commencement of truly terrestrial flora, but they also set the limit to which saltmarsh can colonise seawards, into areas usually vegetated with mangroves or seagrasses. Numerous factors influence the extent of tidal wetlands and tidal inundation regimes, which can vary by coastal setting and estuary type, through controls on intra-estuary geomorphology and modification of the open-coast tide[159]. The landward extent of coastal saltmarsh is largely determined by the penetration of very high, but rare, tides, such as spring high tides or the Highest Astronomic Tide (HAT), whereas the seaward extent is determined by the depth, duration and periodicity of routine (daily) tidal submergence, conflated by the intensity and frequency of mechanical disturbance due to coastal processes (i.e. tidal or wave action). The landward fringe of coastal saltmarsh sets the boundary between variously inundation- and salt-tolerant (or intolerant) woody vegetation such as Swamp Paperbark (*Melaleuca ericifolia*) or Swamp She-oak (*Casuarina glauca*), and truly terrestrial vegetation commonly dominated by genera such as Eucalyptus or Banksia[160]. Tidal inundation also influences the distribution of different plant taxa within a coastal saltmarsh. The 'upper' saltmarsh which receives less frequent tidal inundation and is often hypersaline; and the 'lower' saltmarsh, which is considerably wetter are characterised by different species.

Freshwater inputs

As well as periodic inundation by seawater, saltmarshes are exposed to episodic inundation by freshwater directly through rainfall and indirectly through terrestrial runoff, groundwater inflows, or periodic inundation from flood-swollen rivers that discharge into estuaries. These freshwater flows are vital for sexual recruitment by saltmarsh plants and for the delimitation of coastal saltmarsh from other types of wetland on saline coastal soils. In Victoria, for example, *Juncus kraussii* tends to occur only where there are permanent freshwater seeps in estuarine settings; *Phragmites australis* similarly grows most vigorously where the full salinity of seawater is moderated to various degrees by the presence of fresh water[161]. Excessive freshwater inputs, however, which can occur along urbanised shorelines with large areas of impervious surfaces or result from catchment management practices on agricultural land, pose a hazard to saltmarsh vegetation. For example, it can cause a shift in species composition from more salt-tolerant to less salt-tolerant saltmarsh taxa, or even complete conversion of saltmarsh to brackish-water wetlands. A change in the structural characteristics of saltmarsh, or its degradation will impact the delivery of the coastal defence service due to a change in the vegetation characteristics (see Performance Factors for Hazard Risk Reduction below).

Wave exposure

Saltmarshes thrive on coastlines that are generally sheltered, where they are very effective at stabilising the shoreline. They can also be resilient to periodic extreme events (e.g., hurricanes in a US-based study[36]), however, where wave energy becomes persistently high, this can overwhelm the ability of the vegetation to maintain the shoreline[162].

Approaches for hazard risk reduction

Most of our knowledge on the restoration of saltmarshes comes from New South Wales, with a distinct lack of management information for other parts of Australia (see Knight, 2018[163] for a review of saltmarsh restoration to date). A single approach (Table 2.4) will not always be adequate for rejuvenation of coastal saltmarsh as part of a nature-based strategy to protect shorelines against erosion. Hydrological restoration, while arguably not always sufficient by itself, is a necessary precursor to effective restoration or rehabilitation in many cases. Active planting will be required where the original vegetation cover has been lost or seriously degraded.

Rehabilitation

Exotic plant species are a threat to the integrity of existing saltmarshes and have been a focus of their rehabilitation. In temperate Australian saltmarsh these species include, Tall Wheat-grass *Lophopyrum ponticum*, Groundsel *Baccharis halimifolia*, Slender Celery *Cyclospermum leptophyllum*, Sea Barley-grass *Hordeum marinum*, Pennywort *Hydrocotyle bonariensis*, Spiny Rush *Juncus acutus*, Rock Sea Lavender and Sicilian Sea Lavender *Limonium binervosurn* and *Limonium hyblaeum*, African boxthorn *Lycium ferocissimum*, Buck's Horn Plantain *Plantago coronopus*, grasses such as *Parapholis incurva and Polypogon monspeliensis* and the daisy *Aster subulatus*, which are serious weeds on either localised or widespread scales. Control measures for exotic saltmarsh weeds usually centre on (1) application of herbicides such as Fluazifop-P® (e.g., for *Spartina*); (2) physical removal (e.g., digging out Spiny Rush), (3) the introduction of grazing by domestic stock (e.g., for Tall Wheat-grass); and (4) hydrological manipulations, especially the re-introduction of tidal flushing[164]. Success can be variable and is very likely to be highly site-specific and require repeated management interventions.

Exclusion fences to remove vehicles, domestic or feral animals have also been a key step in limiting mechanical impacts to the rehabilitated saltmarsh.

Table 2.4. A summary of the interventions for risk reduction using saltmarshes. × = no; ✓ = yes; ? = information not known. The costs given are minimum – maximum.

		INVASIVE VEGETATION CONTROL, EXCLUSION FENCING	HYDROLOGICAL RESTORATION	PLANTING	HYBRID APPROACHES
LIMITATIONS TO ESTABLISHMENT	**Overcomes hydrological limitation**	×	✓	×	✓
	Overcomes propagule limitation	×	✓	✓	×
	Overcomes time limitation	×	×	×	✓
	Effective against top-down drivers*	✓	×	×	×
	Effective against bottom-up drivers*	✓	✓	×	✓
APPLICATION	**Scalable to large areas (> 1 km^2)**	✓	✓	✓	✓
	High technical expertise required	×	✓	×	✓
	History of use (for restoration)	✓	✓	✓	×
	History of use (for risk reduction)	×	×	✓	✓
	Time to effectiveness (yrs)	?	10 - 25	5 - 10	Immediate
	Set up cost (AUD m^{-2})	‡<1 - 8	?	§<1 - 3	136 - 940†
	Maintenance required	✓	×	✓	✓

*top-down drivers include grazing, trampling; bottom-up drivers include competition, hydrology; †costs per linear metre; ‡costs based on weeding: §some of these estimates include weeding with revegetation. Refer to Knight (2018) for a range of project costs[163].

Hydrological restoration

Coastal saltmarshes have frequently been the subject of severe hydrological alteration through the addition of structures for tidal control, or to protect infrastructure from erosion and flooding, such as flood gates or seawalls, or the placement of roads or walkways. Given the frequency with which the original hydrology has been altered, restoration or rehabilitation of coastal saltmarsh in Australia frequently centres on returning natural patterns of tidal inundation to tidally alienated saltmarshes. These interventions can be made on a large scale (e.g., through the removal of bunds or of floodgates) or through smaller-scale or simpler means (e.g., placing culverts under roads or pathways that dissect and fragment coastal saltmarsh[165]).

Hydrological restoration may also involve infilling or levelling of damaged saltmarsh or altered foreshores to create the appropriate elevation for saltmarsh to colonise. Considerations with this approach include: the fate of existing topsoil and vegetation, the type and source of fill material, fill level, and the methods for filling and levelling. A previous project (Tweed Heads, NSW[166]) used elevations that would exclude mangrove encroachment and allow for fill settlement, the infill was sourced locally and the topsoil preserved to place back on top of the new infill surface.

Saltmarsh planting

Planting will be required if the original halophytic vegetation has been removed and there are no nearby sources to replenish the area via floating seeds brought into the subject area on the tides or by passing waterbirds, or by plant fragments washing in from outside, which then establish within the target area). Direct seeding is the most efficient method for planting saltmarsh, and seeds can be collected from multiple donor plants to maximise genetic diversity. Other methods include transplanting saltmarsh plugs from healthy donor sites, however, a downside to this method is the impact on these donor sites. Plant cuttings is one way to avoid this, and is a method that can be used year-round, but is also more time-consuming and labour intensive than direct seeding, which does not require planting or propagation. Regardless of the method, general considerations for saltmarsh planting include, plant provenance and origin, dimensions, conditions, collection, storage and transportation, the method for replanting, as well as pre- and post-planting requirements. Saltmarshes will likely require regular watering post-planting during establishment, depending on the rainfall during that time. In some cases, sediment replenishment (e.g. by seagrass wrack) may also be required to aid successful re-establishment of saltmarsh vegetation[167].

Hybrid approaches

In areas where an increase in wave energy or water flows has led to the degradation of saltmarsh, or is preventing saltmarsh establishment, hybrid approaches have been used. These have often used engineered bank stabilisation structures that can create a suitable environment for saltmarsh establishment. Rock sills, with gaps that allow for tidal inundation of saltmarsh behind them are a common approach employed in the United States[52]. Analogous to this, rock fillets have been widely employed in New South Wales for bank stabilisation (see Case study: Rock fillets with mangroves) and where mangroves often naturally recruit, resulting in a re-profiling of steep estuary banks that allows for the colonisation of saltmarsh and salt-tolerant trees to establish in the upper intertidal.

Smart tide gates[168] are another hybrid approach; these allow for tidal inundation and flooding, but are closed when the tide reaches a certain height. This combined approach can use the coastal protective capacity of saltmarshes during appropriate conditions, but can rely on the engineered flood defences during extreme events, which may be more effective (although not under all hazard types[36]).

Performance Factors for Hazard Risk Reduction

Coastal saltmarshes have the potential to provide protection against flooding and erosion[154]. The drag exerted by vegetation on the waves leads to attenuation of wave energy (through wave attenuation due to roughness, see Section 1.2) subsequently reducing wave runup levels and the potential for flooding. The presence of saltmarsh vegetation also reduces erosion potential through sediment capture and soil stabilisation (erosion mitigation within ecosystem, see Section 1.2). Some studies have pointed out the potential of emergent ecosystems such as salt marsh to reduce wave setup[26] or storm surge levels[30], although these functions are less well-established.

Vegetation characteristics

Although the vast majority of studies focused on coastal protection considered saltmarsh species in North America or Europe, the efficacy of a marsh in providing protection against coastal hazards relies on a number of generic vegetation characteristics (see Section 1.2, Table 1.1). Shepard *et al*. (2011)[154] conducted a systematic review of studies focused on coastal protection services provided by saltmarshes, providing a comprehensive overview of the governing vegetation characteristics. A key parameter is the cross-shore width of the marsh, i.e., the length over which waves interact with the vegetation. A wider marsh results in more wave attenuation and a larger area in which sediments are stabilised. Saltmarsh is generally most effective in attenuating wave energy for a width larger than 10 m (e.g., wave attenuation rates > 30%), although large attenuation rates have also been observed in some studies for marshes of only a few meters wide[154]. For marsh width > 100 m, the majority of wave energy is attenuated, e.g., > 80% for a marsh of 160 m[169]. It should be noted that most field studies were conducted under relatively low-energy wave conditions. Higher rates of wave attenuation are obtained where there is a larger plant frontal area (combination of plant height and width/diameter that interact with the water), larger density/coverage (number of plants per area), and larger plant stiffness. Saltmarshes that are able to effectively attenuate wave energy also allow for increased sediment stabilisation due to the reduced hydrodynamic forcing on the bottom. In addition, increased root density generally leads to increased protection against erosion[170].

Site characteristics and hydrodynamic conditions

In general, wave attenuation by vegetation is strongly correlated with the incident wave height and to a lesser extent wave period (see Section 1.2, Table 1.1). Larger waves result in larger wave-driven flow velocities, which increases the drag forces acting on the plants and vice versa on the water, leading to increased wave attenuation[171]. However, in some cases larger waves may actually lead to reduced attenuation, for instance when plants bend substantially with the wave motion or are even damaged due to the wave impact. This process is very much case- and species-specific as it depends on the plant stiffness and resistance/strength relative to the incident wave conditions.

In addition to ecosystem drag, waves can attenuate energy due to bottom friction or wave breaking which both depend on the local water depth. Energy attenuation due to bottom friction is usually much smaller than ecosystem drag as the substrate within these systems is generally relatively smooth (e.g. mud). However, depending on the incident wave height, wave breaking can be a major contributor to wave attenuation, particularly at the saltmarsh edge where the local water depth typically reduces over a relatively short distance[169].

Mangroves

Mangroves are vegetated coastal wetlands comprised of trees and shrubs that occur in the high- intertidal zone on low-energy subtropical and tropical coastlines. Estuaries are key mangrove habitats, but they also occur in lagoons, on open coasts and river deltas. There are 72 species in the biome from a range of plant families, which have different tolerances to environmental conditions. Mangroves are particularly important for reducing risk against coastal hazards such as flooding and erosion. They are known to be able to effectively attenuate incident wave energy, slow flood currents and reduce the impact of tsunami waves[172].

In Australia mangroves are found in most parts of the coastline, but are absent from Tasmania, the Great Australian Bight and southern Western Australia, where they occur in only a few estuarine locations (Figure 2.1). Their diversity is very low (single species *Avicennia marina*) in the south of Australia, but increase in more northern sites, reaching maximum diversity in the wet tropics of Queensland (~35 species)[173]. Low and highly seasonal rainfall results in lower floral diversity compared to wetter sites. The extent of mangrove development is strongly influenced by coastal geomorphology. Mangroves are most highly developed in extent, tree size and floral diversity in estuaries where conditions can range from brackish to hypersaline, thereby providing a wide range of ecological niches[173]. Coastal geomorphology varies between temperate and tropical coastlines giving rise to different kinds of mangroves. In the south of Australia, mangroves can form narrow fringes around estuaries formed in drowned river valleys. In the north of Australia where there are large floodplains at low elevations and extensive tidal creek systems, mangroves fringe the edges of tidal creeks and rivers, forming extensive stands in some locations.

Left: Mangrove and saltmarsh habitats in the lower Richmond River estuary near Ballina, NSW © Patrick Dwyer; Right: Mangroves in the Hinchinbrook Channel, QLD © Fernanda Adame.

Mangroves have been removed to accommodate coastal development in many locations, but in other locations they are expanding due to the increased availability of fine sediments, derived from erosion of catchments with agriculture, which are increasing areas of tidal mudflats at or above mean sea level and thus are suitable for mangroves to colonise[174]. In some locations mangroves are expanding landward into saltmarsh habitat as sea level rises and prolonged drought causes subsidence of sediments, both which result in higher frequency of tidal inundation that favours mangrove establishment and growth[175].

Habitat requirements for mangroves

While there are general factors affecting the distribution of mangroves, different species have varying tolerances to environmental conditions (e.g., salinity, duration of inundation, waves) that are reflected in the zonation patterns of species occurrences observed in natural mangroves. In Darwin Harbour, for example, *Sonneratia alba* is tolerant of high levels of inundation and typically occurs low in the intertidal, while *A. marina* is tolerant of high levels of salinity and occurs in landward positions. Species tolerances of salinity and/or inundation may vary depending on additional environmental factors (e.g., nutrient availability, temperature).

Temperature

Low temperature is a primary control on the latitudinal limits of mangroves globally. They are constrained by sensitivity to frost, and the southernmost populations occur at approximately 37-38°S latitude in Australia and New Zealand[174, 175]. Increases in temperature have facilitated mangrove encroachment into landward saltmarsh habitats in some of these locations[176]. Different species have different tolerances of temperature and humidity[177] and projects seeking to rehabilitate or create mangrove habitat should select species that are suitable for prevailing conditions.

Inundation regime

Mangroves are sensitive to inundation frequency and intensity. They grow approximately between mean sea level and the level of the highest astronomical tide (the upper intertidal zone). They generally grow poorly if submerged more than half of the time (with the exception of some species) or not submerged sufficiently frequently where hypersalinity of soils can develop which restricts growth of many species[178]. Mangroves may occupy a small portion of this intertidal range if, for example, hypersaline conditions occur in the high intertidal zone because of high levels of evaporation and limited groundwater availability (e.g. in the arid zone) or wave energy is high, preventing recruitment in the lower intertidal zone. The loss of mangroves in many areas along the coast has been due to a modification of tidal inundation, for example through building levees or flood gates associated with drainage of land for agriculture, which is a potentially reversible process to restore wetlands.

Salinity

Most mangroves grow rapidly at approximately 25% of the salt concentration of seawater (~8 PSU) with growth rates often declining at higher salinities, although species differ in their maximum growth rates[179]. Mangroves use groundwater when available which stimulates growth rates and forest development[180] and many species may have the capacity to harvest water from dew and rainfall[181, 182]. Although some species can survive hypersaline conditions (salt concentrations > than seawater), they are not tolerant of dry soils.

Wave exposure

The establishment of mangrove propagules and seedlings is sensitive to waves as they can cause failure of root systems that anchor seedlings in soft sediments[183]. Seedlings can recruit in wave affected shores if there is a long enough “window of opportunity” of low wave energy to allow growth to a size that can withstand high levels of wave attack. Exposure to high wave energy and water flow rates can dislodge propagules and seedlings during early phases of mangrove development and can erode sediments from around roots in mature trees leading to toppling and shoreline retreat. Thus, on open coasts mangroves are confined to lower energy environments that are often protected by extensive mud, sand flats, or coral reefs. Adult mangroves provide protection from hydrodynamic stress to establishing seedlings. Where mangroves have been lost from an area, an increase in hydrodynamic energy can support an alternative stable mudflat state, which can be challenging to re-establish mangroves[66]. In these areas, temporary protection of new recruits may aid in seedling establishment, via individual (e.g. seedling guards, concrete pods) or larger scale (e.g. sand bags, pile fields) structures/measures. Although long term sustainability of the mangrove is dependent on maintaining suitable conditions, which improve theoretically as the forest matures and re-establishes the local protection from waves that was lost when clear-felled. Accelerating sea level rise will result in the incidence of higher wave energy on coasts in the future, which may damage mangrove stands.

Sediment type and supply

Many mangrove species grow most rapidly in fine sediments and can colonise fluid muds if they occur at a sea level within an appropriate energy regime[184]. Mangroves can colonise sands (e.g. Moreton Bay) and carbonate sediments (e.g. coral cays), where they often lead to accumulation of carbon rich sedimentary deposits[185]. On sand and carbonate sediments mangroves can be limited by nutrient availability, which can be low in these sediment types[186].

Large deposits of sediment during storm events can be fatal to some species if the aerial roots are covered and anoxia in the root zone develops[187], and can also decimate restoration projects by smothering plantings. Storm deposits, however, can also contribute to accretion and nutrient delivery and support long term sustainability of mangroves[188, 189].

Approaches for hazard risk reduction

A common cause of mangrove loss or their inability to recover from extreme events is because hydrological conditions have been altered, either through human interventions or due to natural processes. This can include reduced tidal inundation[190]. Increases in wave attack following mangrove removal can lead to erosion, which results in feedback processes that favour mudflats rather than a vegetated state[191]. Rebuilding of mangroves, therefore, often requires reinstatement or restoration of suitable hydrological regimes, either with planting of propagules or with regeneration of mangroves from natural recruitment (Table 2.5). The use of natural mangrove recruitment is the favoured approach in restoring mangroves[192], however, requires a reliable source of propagules and a 'window of opportunity'[183] for successful recruitment.

Hydrological restoration

Anthropogenic activities such as building of levees and seawalls for conversion of coastal floodplains for agriculture, addition of sediment for land reclamation, or erosion of sediment (e.g., through sand mining or severe storm events) can create an inundation regime unsuitable for mangrove survival. This, however, may be reversible in some cases through management actions such as the deliberate breaching of dikes or levees, or excavation or infill of sediments to create habitat with appropriate tidal inundation to support mangrove growth. Where dikes or levees have been built to restrict tidal inundation, these can be removed or regraded. However, where full removal is not possible either logistically or economically, a breach in the wall, which creates a channel akin to a natural tidal creek can be all that is required[192]. These channels use the natural tidal forces to maintain the water-course and prevent them from filling up with sediment. Therefore, the number of breaches made needs to take into account the hydrodynamics of a particular site[193].

Where the addition of too much sediment has occurred at a site, the elevation can be outside of the tidal range, preventing mangroves from establishing. In these cases, the removal of sediment has been successful in restoring the natural hydrology for mangroves to re-establish[192]. If a site has low substrate elevations, then fill can be added to the site. However, if the cause of the low elevation is driven by high wave energy and erosion, then it can be difficult, or impossible to maintain suitable conditions for mangrove growth. In these cases, it may not be possible to use mangroves for coastal defence in that area, or hybrid approaches can be considered.

Mangrove planting

It is more cost-effective to restore a site that can be naturally colonised by mangroves. For this to occur the site needs to be hydrologically connected to a natural mangrove forest for the propagules or seeds to be carried to the site. If natural recruitment cannot occur, then planting of mangroves has been commonly used in restoration, although high failure rates have been observed where underlying conditions are not suitable[194].

Mangrove planting commonly uses direct seeding techniques (or assisted dispersal) or planting of nursery-reared seedlings, however, there have also been limited trials using transplanting techniques. Many mangrove species have buoyant propagules rather than true seeds. Propagules are seedlings that are germinated while still attached to the maternal plant (called vivipary), and thus they are often large, have no dormancy, have green, photosynthetic tissues and are adapted for dispersal in water and for rapid establishment. In a direct seeding approach, propagules are collected during the reproductive season of the mangroves and planted directly at the site following removal of the pericarp that encloses the developing propagule, although this does not necessarily enhance seedling performance[195]. Propagules of species in the Rhizophoraceae are easily collected and planted and thus often planted in huge numbers in restoration and afforestation projects, although these have been prone to failure when appropriate environmental conditions are not considered[194]. *Avicennia marina* (grey mangrove), which is dominant in Australia's southern regions also produces propagules. These are round in shape and can be attached to stakes to secure them in place while they grow the roots that secure them in the sediment[196]. Other common species (e.g., those in the genus *Sonneratia*) have true seeds that are not buoyant. An alternative approach is raising propagules in a nursery, where they are grown for between 3-12 months before the young seedlings are planted at a site. Direct seeding or assisted dispersal is more cost-effective over nursery-reared seedlings, as an off-site facility is not required. However, the survival of larger seedlings may be greater. Mangrove nurseries and planting can provide opportunities for capacity building, community development and publicity. If planting is used, then use of appropriate species is important. Having a clear understanding of local species and how they are distributed over the intertidal zone can guide species selection.

Hybrid approaches

In areas that are prone to erosion, or where the hydrodynamic environment has become unfavourable for the restoration of mangroves, hybrid approaches have been used to promote accretion of sediments, which then facilitate mangrove establishment. A hybrid approach uses an engineered structure that reduces hydrodynamic energy and stabilises the sediment to create an environment that naturally recruited or planted mangroves can survive[197]. Experimental projects using brush fencing to attenuate waves and trap sediments have been deployed in Indonesia[197] and the Mekong Delta[198]. A history of this approach in Australia is using rock fillets in New South Wales (See Box 2.2, Case study: Rock fillets with mangroves). Further, concrete mangrove planters are being trialled in Victoria. In other areas globally, techniques have included temporary breakwaters such as PVC pile fields, recycled tyres, or bamboo. Most of these techniques are experimental, and therefore need to be supported by rigorous monitoring programs to inform their effective use, cost effectiveness, delivery of additional ecosystem services and sustainability over the long term.

Table 2.5. A summary of the interventions for risk reduction using mangroves. × = no; ✓ = yes; ? = information not known. The costs given are minimum – maximum, or a mean.

		HYDROLOGICAL RESTORATION (WITH NATURAL REGENERATION)	PLANTING: DIRECT SEEDING	PLANTING: NURSERY REARED	HYBRID APPROACHES
LIMITATIONS TO ESTABLISHMENT	**Overcomes hydrological limitation**	✓	×	×	✓
	Overcomes propagule limitation	×	✓	✓	×
	Overcomes time limitation	×	×	×	✓
	Effective against top-down drivers*	×	×	×	×
	Effective against bottom-up drivers*	✓	×	×	✓
APPLICATION	**Scalable to large areas (> 1 km^2)**	✓	✓	✓	✓
	High technical expertise required	✓	×	×	✓
	History of use (for restoration)	✓	✓	✓	×
	History of use (for risk reduction)	✓	✓	✓	✓
	Time to effectiveness (yrs)	10 - 25	10 - 25	10 - 25	Immediate
	Set up cost (AUD m^{-2})	?	3.10‡	6.10‡	136 - 940†
	Maintenance required	×	✓	✓	✓

*top-down drivers include grazing, trampling; bottom-up drivers include competition, hydrology; ‡costs per seed/seedling; †costs per linear metre

Performance factors for hazard risk reduction

The ability of mangroves to provide coastal protection is now widely established[199]. They are able to effectively attenuate wave energy through drag (wave attenuation due to roughness, see Section 1.2), thereby reducing wave runup and the potential for coastal flooding. Mangrove forests can also reduce the potential for erosion by promoting sediment settling[200] through creating a low-energy environment, as well as soil stabilisation through their subsurface roots[201]. While most studies have addressed the capacity of mangroves to attenuate sea-swell wave energy, some studies found mangrove forests may also reduce (to a lesser extent) infragravity wave energy[25], surge-induced nearshore water levels and tsunamis[202] (see Section 1.2).

Vegetation characteristics

The ability of mangroves to attenuate wave energy is largely driven by the cross-shore width of the forest (i.e., length over which the waves interact with mangrove trees). For instance, based on satellite imagery spanning a 13-year period, Phan et al. (2015) studied various locations along the southern coast of the Mekong Delta and found a strong relationship between mangrove forest width and coastline stability. They found that depending on the location a width of 30 to 250 m with an average of 140 m was required for the coast to remain stable. For specific locations, however, they found substantial rates of erosion even for relatively wide mangrove forest (> 200 m), which were attributed to the presence of sea dikes located directly landward of the mangroves. The authors hypothesised that these sea dikes allow for reflection of (in particular) infragravity waves leading to greater potential for erosion. Higher rates of wave attenuation are expected for mangroves with relatively large frontal area interacting with the waves, which is function of the root biomass, trunk diameter and tree canopy in case of (near) submergence. The density or coverage of mangrove trees (number of trees per area) also directly affects the rate of wave attenuation[203]. By attenuating wave energy, mangroves create a calm environment for sediments to settle, and reduce bottom stress and consequently the potential for erosion. Increased density of subsurface roots generally leads to increased protection against erosion.

Site characteristics and hydrodynamic conditions

Wave attenuation by vegetation drag is strongly correlated with the incident wave height and to a lesser extent wave period (see Section 1.2, Table 1.1). Larger waves result in larger wave-driven flow velocities, which increases the drag forces acting on the mangrove trees and vice versa on the water, leading to increased wave attenuation[203]. In extreme cases such as cyclones or tsunamis, the wave or wind impact may be strong enough to damage or uproot individual trees leading to reduced coastal protection capacity. Studies that specifically assess the damage on mangroves during extreme events are sparse and report case-specific results. For instance, Kamthonkiat *et al.* (2011)[204] reported approximately 5% loss of mangroves in their study area in Thailand as a result of the 2004 Indian Ocean tsunami. Primavera *et al.* (2016)[205] found a wide range of damage levels due to cyclone Haiyan across the mangrove areas included in their study in Philippines, with up to 86% of trees that died in areas that were in the direct path of the cyclone.

In addition to ecosystem drag, waves can attenuate energy due to bottom friction or wave breaking, which both depend on the local water depth. Energy attenuation due to bottom friction is usually much smaller than ecosystem drag as the bottom within these systems is generally relatively smooth (e.g., mud). However, depending on the incident wave height, wave breaking can be a major contributor to wave attenuation.

Box 2.2 Case study: Rock fillets with mangroves, New South Wales[206]

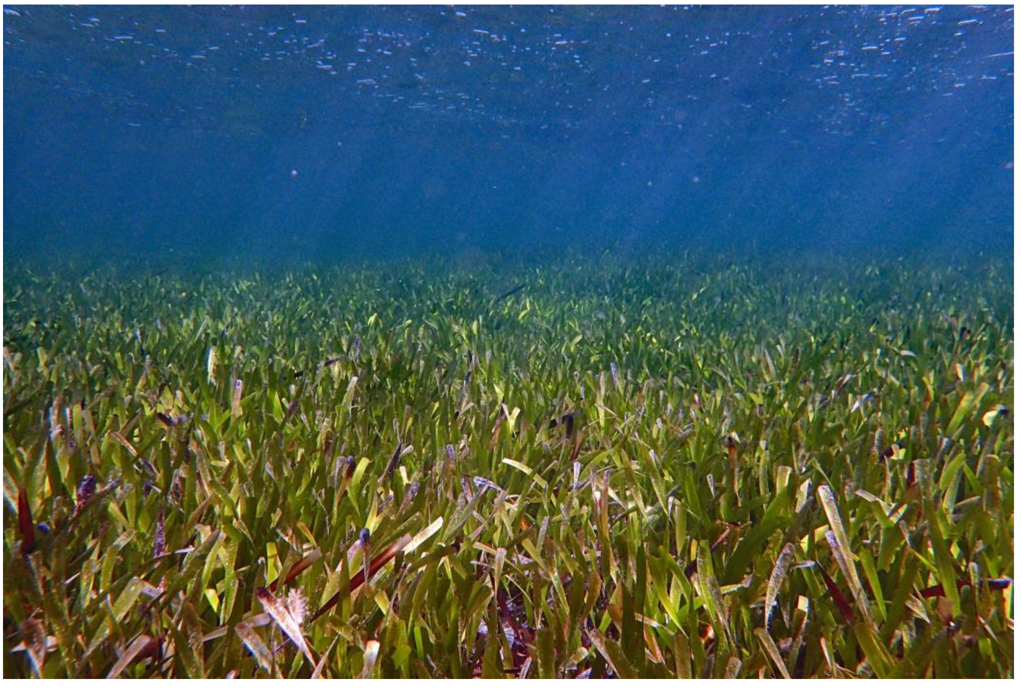
Rock fillets have been used as an estuary bank stabilisation method in New South Wales since 2000, where the first fillets were installed at Dumaresq Island. Since, they have been deployed at multiple sites and estuaries, especially on the northern NSW coast. Rock fillets are energy dissipating structures that provide immediate toe protection to an eroding estuary bank. They are often constructed to mean high water level and are made of piled rocks (300 – 700 mm diameter) on geofabric, however, wood structures have also been used. Rock fillets are keyed into the upstream estuary bank, with a gap at the downstream end that allows for tidal flushing, fish passage and the natural recruitment of mangroves. The sheltered area between the rock fillet and bank stabilises and promotes accretion of sediment. This provides a suitable environment for the natural establishment of mangroves, where the erosion rates were previously too high for survival. Rock fillets are suitable where there is a wide (at least 5 m) shallow intertidal bench in front of the bank, as well as some space landward of the bank for the mangrove fringe to widen over time as the bank re-profiles. This approach has been considered successful in erosion control, while recovering some of the natural functioning of the shoreline. In many cases this approach has been supported by the fencing of boundary lines to prevent livestock from trampling or eating the mangroves.

Rock fillet, with first year mangrove recruits, Ballina NSW

© Rebecca Morris

Seagrasses

Seagrasses are a group of marine angiosperms that are present shallow water habitats of all continents except Antarctica. As angiosperms, seagrasses have true root systems that act to stabilise sediments, as well as producing fruits and flowers for sexual reproduction. Seagrasses can also expand spatially through asexual reproduction via rhizome extension. Seagrasses are a collective group spanning at least three families of flowering plants that have returned to the ocean. Seagrass species can also be functionally classified into three groups: colonising, opportunistic and persistent. Classification within these groups is dependent on factors such as life-history strategies, habitat and meadow form, which all have important implications for management[207]. Colonising seagrasses (e.g., *Halophila*, *Halodule*) are much smaller, and tend to rely on a life-history strategy centred around fast recovery. Persistent seagrasses (e.g., *Posidonia, Thalassia)* are large seagrasses that can persist through adverse conditions by relying on stored carbohydrates, but have slow recovery rates after loss. Opportunistic species (e.g., *Zostera*, *Amphibolis*) tend to be somewhere in the middle of this spectrum, showing varying rates of recovery and resilience. These functional groupings have other key differences, including their propensity for sexual reproduction (generally higher in colonising seagrasses) or asexual propagation (generally higher in persistent seagrasses) and traits such as shoot turnover and seed dormancy. Seagrass meadows are major coastal habitats present across the Australian coastline, in both tropical and temperate locations (Figure 2.1). As well as sediment stabilisation, they contribute to coastal protection through increasing bed roughness leading to wave attenuation. Australia is home to 22 species of seagrasses[208], making it one of the global hotspots for seagrass diversity.



Posidonia australis bed, Western Australia © Rachel Austin.

Habitat Requirements for Seagrasses

Light

Seagrasses can exist down to 40m depth but are typically present in shallow waters less than 15m deep. This is due to their relatively high light requirements (10% of surface irradiance, compared with ~1% in most algal species), with seagrasses requiring adequate photosynthetically active radiation for photosynthesis and growth. This leads to seagrasses being widely distributed around coastal areas, given the strong relationship between water depth and light attenuation. This light requirement also leaves seagrasses vulnerable to changes in water quality. Increases in coastal eutrophication that decrease light availability have historically been a driver of seagrass loss in Australia[209]. In addition, other factors that reduce light availability can be important in some areas such as increased sediment inputs and overgrowth by macroalgae can negatively impact seagrasses and even lead to declines. Seagrass loss occurs when long-term light availability falls below minimum light requirement levels, or through interactions between decreased light availability and other environmental factors, such as water temperature[210] or sediment characteristics[211, 212]. However, seagrasses can also positively impact their surrounding light environment through limiting sediment resuspension, which provides a more favourable light environment for seagrass and other photosynthetic organisms[213].

Salinity

As the only fully submerged marine angiosperms, seagrasses have developed a range of adaptations that allow them to survive in submerged environments with relatively high salinities. These include a loss of stomata, the presence of aerenchyma, and reduction of cuticle. Seagrasses differ from other aquatic plants in being able to thrive in oceanic salinities but can be also found in brackish waters in estuaries, and hypersaline waters in reverse estuaries such as Shark Bay, WA and Spencer Gulf, SA. Often, strong gradients in salinity can also interact with changes in nutrient availability, with both influencing productivity[214]. Changes in salinity can also be an important cue for seagrass seed germination, meaning that local salinity fluctuations may have to be considered in the context of seed-based establishment of seagrass.

Sedimentology and hydrodynamics

Seagrasses thrive in coastal ecosystems with a variety of differing sediment and hydrological conditions. Seagrasses are usually found growing on muddy to sandy sediments, though some genera such as *Thalassodendron* and *Amphibolis* can grow in coarser sediments. Unlike macroalgae, seagrasses can directly access nutrients from sediments, and therefore often outcompete algae in waters with extremely low nutrient concentrations. Conversely, under eutrophic conditions macroalgae can take up excess nutrients and grow rapidly, outcompeting seagrasses and smothering them. Seagrasses typically grow in calm waters but can persist in exposed areas with high hydrodynamic flows with their root systems contributing to anchorage. Hydrodynamic flows play a major role in shaping seagrass ecosystems. For example, strong storms can open up gaps in seagrass canopies, decreasing total seagrass cover, but can also replenish nutrient delivery into remaining meadows. Wind and currents can also increase seagrass seed or asexual propagule dispersal into unvegetated habitats and could contribute to colonisation of these habitats. Some seagrasses (e.g., *Amphibolis antarctica)* favour hydrodynamically active areas, and have adaptations that allow them to survive under such conditions.

Approaches for hazard risk reduction

Given that a change in light availability through eutrophication and suspended sediments is one of the primary causes of seagrass decline, it is important to ensure that sites have either been remediated or have suitable light prior to attempting to use seagrass as a nature-based method. This can include using available physical data, historical seagrass distribution data, species distribution models and ecological niche factor analysis to determine suitability of habitat for restoration projects[215]. Where the environmental conditions support seagrass meadows, restoration has been shown to be feasible and cost-effective in an Australian context[216]. This is usually done through active planting methods (Table 2.6).

Active restoration of seagrass

Rhizome fragments

Rhizome fragments are overwhelmingly the dominant planting unit used in seagrass restoration projects globally. This method involves collecting rhizome fragments (usually with 3-8 shoots, including an apical meristem) from the donor meadow, transporting in seawater to the recipient site, and planting at an appropriate depth in sediments. Often, rhizome fragments are anchored into sediments using items such as weights, staples or pegs. This keeps the fragments anchored to the seafloor while new root growth is established, preventing loss through hydrodynamic movement and dislodgement, a major contributor to seagrass loss in the early stages of restoration[217]. The efficacy of anchoring appears to be species and location specific, though Australian species such as *Posidonia australis* have benefited from anchorage, with unanchored units being removed completely by water movement[218]. Though rhizome fragments have traditionally been used as the planting unit for restoration trials, they tend to be relatively expensive on a cost-per-transplant basis, given the high level of handling (and expert time) required to carry out transplanting of fragments. However, when resourced to an appropriate level these projects can be successful, given the positive relationship between restoration scale and survivorship[217].

Seagrass cores

Seagrass cores (or sods/plugs) have also been trialled in restoration projects. Seagrass cores contain seagrass fragments that also have intact native sediments around roots and rhizomes. The inclusion of intact sediment in cores improves the initial survival of plants by limiting stress and increasing anchoring of the plant[217]. Restoration with seagrass cores tends to be more time-consuming and expensive than other methods, therefore is not as widely adopted as other methods, although it has been used in Western Australia[219].

Seed based restoration

Seed based restoration is increasingly being viewed as a method to restore relatively large areas of degraded seagrass meadows. This approach takes advantage of the natural capacity of seagrasses to produce seeds which can disperse and colonise new areas[220]. For restoration of seagrasses with direct developing seeds (e.g. *Posidonia*), seagrass seeds can be collected and transported to target areas for restoration, while other species (e.g. *Zostera*) that produce seeds at the end of the growing season require seed storage after collection until growing conditions improve. Seagrass seeds vary widely in their morphological characteristics, dormancy and germination periods. Seeds vary in size from <1mm in *Halophila* species to >1cm in *E. acoroides*, while some seagrasses have periods of distinct seed dormancy while others do not. For example, *Halophila decipiens* produces negatively buoyant seeds that are buried in sediments and remain dormant until environmental conditions are suitable, after which they undergo germination[221, 222]. In contrast, *Posidonia australis* produces large, buoyant fruits on inflorescences that are released when ripe. These fruit float, and are moved by wind and waves, before discharging a negatively buoyant seed that settles in the sediment[223]. Timing of seed production and release is also species-specific and depends on local environmental conditions. When information is present on seed biology, seagrass seeds represent an ideal planting unit for restoration work. Indeed, several projects have utilised seagrass seeds for successful restoration outcomes[216], and the largest successful seagrass restoration involved using 70 million *Zostera marina* seeds to restore an area of 3612 ha of seagrass[224]. Though the per unit survival of seeds is lower than using rhizome fragments, the number that can be quickly collected and deployed means that this approach may be more cost-effective for areas that require sizable seagrass meadows for coastal defence outcomes.

Hybrid approaches

While still under development, research has shown that seagrass establishment can be increased using hybrid techniques that suppress waves or sediment mobility[225]. These structures are preferably biodegradable, and past applications have included the use of BESE-elements® starch mesh[225] and coir logs (e.g., used by the Estuary Care Foundation, South Australia). The placement of these structures can be used in combination with active restoration approaches. Similarly, the deployment of offshore structures, such as oyster reefs can also have positive effects on seagrass cover in the lee of the reef, which could be achieved through an improvement in water quality and/or wave attenuation[226]. Integrated habitat restoration that supports the establishment of multiple habitats has the potential for even greater hazard risk reduction, as well as multiple co-benefits (see Section: Multi-habitat Restoration).

Table 2.6. A summary of the interventions for risk reduction using seagrasses. × = no; ✓ = yes; ? = information not known. The costs given are mean or minimum – maximum.

		RHIZOME FRAGMENTS	SEED BASED RESTORATION	HYBRID APPROACHES
LIMITATIONS TO ESTABLISHMENT	**Overcomes propagule limitation**	✓	✓	✓
	Effective against top-down drivers*	×	×	×
	Effective against bottom-up drivers*	×	×	×
	Addition of resilient genotypes	✓	✓	✓
	Propagation of resilient genotypes	✓	✓	✓
APPLICATION	**Scalable to large areas (> 1 km^2)**	×	✓	?
	High technical expertise required	✓	×	✓
	History of use (for restoration)	✓	✓	×
	History of use (for risk reduction)	×	×	×
	Time to effectiveness (yrs)	1-5	4-10	1-5
	Set up cost (AUD m^{-2})	17.8	3	2.7 – 37.5
	Maintenance required	✓	✓	✓

*top-down drivers include grazing; bottom-up drivers include water quality

Performance Factors for Hazard Risk Reduction

Seagrasses have the potential to provide protection against flooding and erosion[153]. The drag exerted by vegetation on the waves leads to attenuation of wave energy (through wave attenuation due to roughness, see Section 1.2) subsequently reducing wave runup levels and the potential for flooding. The presence of seagrass vegetation also reduces erosion potential through sediment capture and soil stabilisation (erosion mitigation within ecosystem, see Section 1.2).

Vegetation Characteristics

Wave height reductions in seagrass meadows are typically tens of percent per 100 m of wave propagation through the meadow[227, 228]. The capacity of seagrass to create wave attenuation due to its roughness tends to increase with the (cross-shore) width of the bed, the seagrass height relative to the water depth and the seagrass density (often expressed as frontal area per unit volume, which is also dependent on blade morphology) and decreases with canopy flexibility (see below)[229, 230]. Seagrass roughness can significantly reduce the near-bed wave-driven velocity which, in turn, can greatly diminish the stress exerted on the sediment bed[230]. Accordingly, seagrass meadows are typically (but not always) regions of reduced sediment erosion and enhanced sediment deposition[231]. In addition, the network of subsurface seagrass rhizomes increases the critical shear stress required to remobilise sediment[232]. Thus, seagrasses maximise their capacity for sediment trapping by diminishing the actual stress on the sediment bed and increasing the threshold stress required to mobilise it.

Site Characteristics and Hydrodynamic Conditions

Particular aspects of seagrass behaviour may constrain its capacity to provide coastal protection, in particular under extreme conditions. Firstly, its flexibility means that seagrass tend to be pronated in strong flows, exerting less drag and creating less wave attenuation. Secondly, seagrasses in temperate regions senesce in colder months[233] such that, in many Australian coastal regions, seagrass coverage is out-of-phase with storm likelihood[234]. Despite these characteristics, seagrass meadows can indeed provide critical coastal protection during extreme events. As an example, seagrass meadows had a strong buffering effect against the 2004 Boxing Day tsunami in Aceh and Southern Thailand. Across 623 sites that experienced coastal flooding, the flooded area was significantly lowered in the presence of seagrass beds[235].

Similar to saltmarshes and mangroves, energy attenuation due to bottom friction is usually much smaller than ecosystem drag as the bottom within these systems is generally relatively smooth (e.g., mud). However, depending on the incident wave height, wave breaking can be a major contributor to wave attenuation.

Kelp forests

Kelps are large brown algae from the orders Laminariales and Fucales[236, 237]. Worldwide, kelps are the dominant habitat-forming species on shallow temperate and subpolar rocky reefs [238] and are found in all continents except Antarctica [239]. Kelp forests are comprised of a collection of seaweeds, which provide food and structure to a myriad of other benthic and pelagic organisms[237]. The diverse morphologies of kelp can be split into three broad guilds: floating canopy (large species with fronds at or near the surface); stipitate (erect understorey where fronds are supported by a stalk or 'stipe' above the understorey) and prostrate canopy (fronds lie on or immediately above the substratum).

In Australia, kelp species are found in both temperate intertidal and subtidal habitats, with their depth limited by light and substratum availability[240]. Kelp forests cover >8,000 km of the coastline (Figure 2.1)[236], and their distribution ranges from southern Queensland throughout southern Australia to Kalbarra in Western Australia (Figure 2.1). The dominant kelp species vary in size from less than one meter to over 40 m in length (Table 2.7)[241], with kelp forests in Australia comprised of mixed stands of diverse fucalean kelp species interspersed with relatively few laminarian kelp species[242].

In terms of coastal defence, kelp differs from the other hard substratum species (i.e., corals and shellfish), in that it is not a reef-forming species. Kelp vegetates a reef, and in doing so interacts with a greater portion of the water column compared to bare reef, which increases drag, with a potential effect on wave transmission[243]. A healthy kelp forest is self-maintaining, however, whether kelp helps to retain the structure of a reef (like corals and shellfish) is unknown. This means that the same degradation of a reef structure (e.g., an artificial reef) may occur over time, even when vegetated by kelp, akin to the design life of a traditional coastal protection structure.

Habitat requirements for kelp forests

The niche occupied by kelps varies across locations, and their establishment in new locations is dependent on the availability of suitable substrata, temperature, light and nutrient (especially nitrogen) levels and wave exposure[244]. The exact requirements of kelps for each of these factors, differs between species and genera (Table 2.7).

Water temperature

Most kelp species occur on hard substrates such as bedrock, boulders, cobbles, or biogenic structures (e.g., mussels or shells)[237] in the cold-water coastal zones (approximately 5 to 20°C) of Australia. The most widespread and abundant species, *E. radiata*, occurs all the way into subtropical waters (Kalbarri in WA, and the border of NSW/Queensland). Kelps can become physiologically stressed, and more susceptible to disease, and mortality in high sea temperatures[237] or during heatwaves[245].

Table 2.7. Habitat-forming kelp species in Australia.

GENUS (ORDER)	DISTRIBUTION	GUILD	THALLUS LENGTH (MM)	HABITAT	WAVE EXPOSURE	TEMP. APPROX. RANGE (^{o}C)	NITROGEN INPUTS	LIGHT (µMOL M^{-2} S^{-1})	RISK ↓ BENEFITS TESTED
Hormosira (Fucales)	0 m, Southern WA, to Northern NSW and around TAS	Prostrate	300	Intertidal	Sheltered	9 – 26 sea >40 air	Low tolerance	2 – full sunlight	×
Carpoglossum (Fucales)	1-40 m, SA, VIC and around TAS	Prostrate	2000	Subtidal	Moderately exposed	Unknown	Unknown	Unknown	×
Cystophora (Fucales)	0-38 m. Northern WA, to Northern NSW and around TAS	Prostrate or stipitate	600-4000	Subtidal	Range of conditions	Unknown	Unknown	Unknown	×
Durvillaea (Fucales)	0-30 m, SA to NSW and around TAS	Prostrate	8000	Lower intertidal	Exposed	9 – 26 sea >30 air	Low tolerance	Unknown	✓
Ecklonia (Laminariales)	0-60 m Northern WA, to Northern NSW and around TAS	Stipitate	2000	Subtidal	Moderately exposed	8 – 24	High tolerance	5 - 50	✓
Lessonia (Laminariales)	0-20 m, VIC and TAS	Prostrate	2000	Subtidal	Exposed	Unknown	Unknown	Unknown	×
Macrocystis (Laminariales)	0-28 m SA, VIC and TAS	Floating	10000-35000	Subtidal	Moderately exposed	4 – 20	High tolerance	6 - 30	✓
Phyllospora (Fucales)	0-20 m, SA to NSW and TAS	Prostrate	3000	Subtidal	Exposed	12 – 23	Low tolerance	Unknown	×
Sargassum (Fucales)	0-50 m Australia wide	Prostrate or stipitate	450-1500	Subtidal	Range of conditions	Unknown	High tolerance	Unknown	×
Scytothalia (Fucales)	0-44 m WA to VIC	Prostrate	2000	Subtidal	Exposed	Unknown	Unknown	Unknown	×
Seirococcus (Fucales)	1-40 m SA, VIC, around TAS	Prostrate	2000	Subtidal	Exposed	Unknown	Unknown	Unknown	×
Undaria (Fucales)	0-10 m VIC, TAS	Stipitate	3000	Subtidal	Moderately exposed	1 - 30	High tolerance	5 - 150	×

Wave exposure

In general, kelp species grow best in moderately exposed waters[246]. In areas with low water motion, the diffusive boundary layer (a thin film of water >1 mm) along the frond limits the capacity of kelps to acquire inorganic carbon and dissolved nutrients and eliminate waste products, which may reduce kelp photosynthesis and growth[247]. Water motion also strongly influences kelp morphology, with numerous adaptations in holdfast, stipe, and blade characteristics that enable them to withstand high-flow environments[248].

However, in extremely high-flow environments and during storms the drag and turbulence can become too high and individuals can be damaged or dislodged[249].

Light and nutrient availability

Kelps require moderate level of light and nutrients. Low levels of either light or nutrients can limit kelp reproductive output, growth, and productivity[250]. However, large inputs of nutrients can favour the growth of epiphytic algae, which can limit the growth and productivity of adult plants[251] and/or promote the growth of other competitors (i.e. turf and opportunistic algae), which act to inhibit the settlement of kelp juveniles[252]. High levels of irradiance can inhibit the photosynthetic activity of adult plants whilst the microscopic stages are often unable to photo-acclimate and experience high levels of mortality[250].

Approaches for hazard risk reduction

Assisted establishment

In many cases, where suitable conditions exist, kelps will naturally recruit to rocky reefs and no further assistance is required. However, where kelp forests have been lost due to anthropogenic stressors (e.g. overgrazing by urchins, oil spills, sedimentation, organic enrichment) further action is often required to remove the stressor(s) leading to their decline[253] (Table 2.8). In many areas, dramatic losses of kelp forests have been linked to overgrazing by herbivores[254]. Notable examples of this include the effects of sea urchins which, through their intense grazing activity, may lead to the eradication of kelps and their replacement with bare rock barren areas[255] or in some cases turfing algae[256]. Negative effects of herbivory on kelp have also been reported where tropical fishes have expanded their range into temperate regions[257]. Herbivore removal has, therefore, been a large focus of establishing kelp beds. Other approaches to restoring kelp include marine protected areas, fishing, and water quality regulation. Multiple studies have demonstrated that kelp forests have successfully established into areas, following the cessation of the disturbance[255, 258]. However, given that kelp spores have limited dispersal (10-100s metres), active approaches to kelp establishment are required if there are no nearby kelp forests.

Active techniques

In locations where there is a lack of natural kelp recruitment or barriers to kelp establishment, more active methods of interventions may be required (Table 2.8).

Transplantation

Transplanting juvenile and/or adult kelps from donor populations to degraded reefs with low propagule supply, can be an important technique for enhancing their natural recruitment[259]. However, the long-term success of this approach is reliant on obtaining sufficient biomass of kelps from a healthy donor population to form self-sustaining populations, which can be environmentally and economically costly. Studies on the effects of transplanting kelps into new areas in Australia have been undertaken at multiple spatial scales with mixed success[260, 261]. The results show that juvenile kelps transplanted to new areas are less likely to form self-sustainable populations than adult kelps. Similarly, kelps transplanted into sites with high wave action and/or high abundances of grazers have realised limited success[260, 261]. One of the most successful programs to establish kelp, has been for *Phyllospora comosa*, in which adult kelps were successfully transplanted into areas of Sydney Harbour where it had gone locally extinct[262]. Within a single generation the transplanted *P. comosa* formed self-sustaining populations[261].

Transplanted *Ecklonia radiata.* © Tristan Graham

Ex situ recruitment enhancement

Kelp transplants can also be cultured in the laboratory via sexual or asexual propagation[263]. In this method, reproductive tissue is collected from kelp plants. This tissue is then used to produce microscopic propagules, which are cultivated to juveniles or released to seed settlement substrates (e.g. strings, clam shells, ceramic plates or gravel) in indoor laboratories and or outdoor nurseries[264]. These substrates are then outplanted at the site of interest. However, the success of ex situ recruitment enhancement is highly dependent on the techniques used to culture the kelps[265], the age of the transplants, with older kelps surviving better than younger kelps[266], the density of juveniles[267], the species of interest and the local site conditions.

Seeding

In some circumstances the establishment of kelp forests can be achieved through seeding. In this technique, reproductive tissues are collected from donor reefs and placed into bags or dispersed into the water column itself. The bags are attached to the bottom at the site of interest[268]. The substratum is then cleaned of potential competitors and/or predators. The seeded kelps are then left to establish natural recruits. However, there have been only a few attempts at using this method, and most studies are limited to testing the effects of seeding in enhancing kelp recruitment, with less than 1 year of monitoring to determine the success of these techniques[269].

Substratum addition

Where there is a lack suitable hard substrata for a kelp forest to establish, artificial reefs may provide suitable surfaces for kelps to settle[270]. Research in Australia has demonstrated that transplanting kelps to artificial reefs has been successful at increasing kelp recruitment and development of a kelp canopy in the short-term (months)[270]. Overall, studies have demonstrated that artificial reefs had varied success in promoting the establishment of kelp, removal of stressors is often key, and in cases where novel substrata is provided with little consideration of other factors, success has been limited/poor[241]. This may be due to the unsuitability of some artificial substrata (i.e. materials, slope or aspect) and/or environmental conditions (light, sedimentation, exposure) in which these reefs are placed. Alternatively, artificial reefs may be colonised in preference by other organisms (e.g. filamentous turf algae, mussels, and barnacles) which limit kelp recruitment[271]. In the right circumstances (i.e. design and site conditions), artificial reefs have the potential not only to promote the establishment of kelps but also to promote beach build-up[272].

Table 2.8. A summary of the interventions for risk reduction using kelp forests. × = no; ✓ = yes; ? = information not known. The costs given are mean or minimum – maximum.

		MPAS, FISHING AND WATER QUALITY REGULATION	HERBIVORE REMOVALS	EX SITU RECRUITMENT ENHANCEMENT	SEEDING	TRANSPLANTATION	SUBSTRATUM ADDITION
LIMITATIONS TO ESTABLISHMENT	**Overcomes substrate limitation**	×	×	×	×	×	✓
	Overcomes propagule limitation	×	×	✓	✓	✓	×
	Effective against top-down drivers*	✓	✓	×	×	×	×
	Effective against bottom-up drivers*	✓	×	×	×	×	×
	Addition of resilient genotypes	×	×	✓	✓	✓	×
	Propagation of resilient genotypes	×	×	✓	×	×	×
APPLICATION	**Scalable to large areas (> 1 km^2)**	✓	✓	?	✓	×	×
	High technical expertise required	×	×	✓	×	×	✓
	History of use (for restoration)	✓	✓	×	×	✓	✓
	History of use (for risk reduction)	×	×	×	×	×	×
	Time to effectiveness (yrs)	5-10	>3	?	?	6-10	Immediate
	Set up cost (AUD m^{-2})	?	3	165	67	9-222	12
	Maintenance required	✓	✓	×	×	×	✓

*top-down drivers include herbivores; bottom-up drivers include water quality

Performance Factors for Hazard Risk Reduction

Kelp forests have the potential to provide protection against erosion and flooding through their three-dimensional structure, which can exert drag on the water column[5] (wave attenuation due to roughness; see Section 1.2). This process in turn can modify sediment transport along the coast[273]. To date, research on the efficacy of kelp forests in buffering waves and/or currents is limited to a few species, the majority of which are not found in Australia (Table 2.7).

Vegetation characteristics

Evidence showing an effect of kelp on wave attenuation is limited and inconsistent, however key parameters that influence efficacy include vegetation height, surface area, density and flexibility (see Section 1.2, Table 1.1)[243]. The effect of vegetation on wave attenuation is greatest when the vegetation occupies a larger proportion of the water column. This means that species which grow taller, in shallow subtidal or intertidal locations, will have the largest effect on wave transmission. However, floating canopy kelp species (e.g., *Macrocystis*; > 40 m length) can reach very large sizes, which span the entire water column. This large size, however, likely allows for these species to move passively with wave motion, minimising drag[243]. This could explain the negligible effects on wave heights that have been observed for large floating canopy species (*Macrocystis)*[274]. However, stipitate kelps (e.g., *Laminaria* or *Ecklonia*) withstand wave energy through increasing strength, rather than flexibility, and therefore have a greater potential to influence surface waves. For example, a laboratory study on *L. hyperborea* (2 m length), at 4 m depth, estimated 50% wave attenuation over a kelp forest, but this became negligible when the kelp occupied less than 20% of the water column (i.e., as water depth increased)[275]. Wave damping also increases with forest width (cross-shore), density and surface area, the last of which can be measured through the leaf area index (the total one-sided leaf area per unit ground surface area[276]; see Section 1.2, Table 1.1).

Site characteristics and hydrodynamic conditions

Through a reduction in water depth, the reefs on which kelp are found can directly influence the wave forces generated during breaking and induce other forms of water motion that can impact on erosion and flooding at the shoreline[243]. In some cases, the site characteristics may act to amplify the wave heights across the kelp forest whereas in other circumstances the opposite can occur[243]. Whether such impacts of reef morphology on wave energy are modified by the presence or absence of kelp requires further study.

The wave conditions can also influence the ability of kelps to alter the hydrodynamics. Previous research has shown that kelps may have a greater effect on wave attenuation under smaller wave heights, as increasing wave height can cause the drag of flexible vegetation to reduce due to the higher velocities and bending forces associated with larger wave[277]. Greater wave attenuation by kelps has been observed to occur for shorter (2–6 s) rather than longer period waves (7–20 s)[275, 278] (see Section 1.2, Table 1.1).

Shellfish Reefs

Shellfish reefs are complex three-dimensional structures created from aggregations of bivalve molluscs, commonly oysters and mussels[279]. They are comprised of both live animals as well as dead shells, which provide the substrate onto which successive generations of bivalves can attach and 'build' the reef structure. Shellfish reefs are distinguished from shellfish beds, typically a single layer of bivalves, by their high vertical relief[80]. The live and dead bivalves that form the foundation of shellfish reefs provide structurally complex habitat that dissipates wave energy through the ecosystem roughness, and shallow bathymetry (wave attenuation by depth-induced breaking; see Section 1.2) and can thus protect shorelines landward of the reefs from wave-driven coastal flooding and erosion[58, 280]. Although some shellfish are capable of forming reefs at great depths, the utility of shellfish reefs to attenuate wave energy is diminished when the reef crest is so deep that waves are transmitted to shore with minimal drag due to ecosystem roughness or wave breaking[281]. For reefs with shallow crests (usually within a few tens of centimetres from mean sea level), oyster reefs can function similarly to low-crested breakwaters in dissipating wave energy[282].

Shellfish reefs occur in intertidal to subtidal waters of bays, estuaries and near-shore coastal waters, at latitudes spanning tropical to temperate climates[279] (Figure 2.1). Within Australia, there are three main native reef-forming species of bivalve[80]: the rock oysters, *Saccostrea glomerata* and *S. cucullata*, and the flat oyster (*Ostrea angasi*). In addition, the non-native Pacific oyster (*Crassostrea gigas*) and blue mussel *Mytilus galloprovincialis*, and the native mussels, *Trichmoya hirsuta* and *Mytilus planulatus* may contribute to shellfish reefs. Other lesser studied species may also be capable of forming three dimensional structures. Shellfish reefs are found in every state and coastal territory in Australia. Shellfish reefs are generally most abundant at intertidal to shallow subtidal elevations, though *O. angasi* can form reefs to depths of tens of metres in some areas[283], and *S. glomerata* historically formed subtidal reefs to eight meters depth[80].

Remnant *Saccostrea glomerata* reefs, NSW. © Francisco Martínez-Baena

Historical records suggest that prior to industrialisation, shellfish reefs were a dominant habitat type in many of Australia's bays and estuaries[80, 284]. However, between the early 1800s and early 1900s, an estimated 85% of reef[285] was lost primarily due to overharvest using destructive fishing practices. Although this fishing pressure has subsequently been removed, shellfish reefs have not naturally recovered[286]. Although oyster reef restoration is very much in its infancy in Australia, pilot projects to date suggest that at many sites the reintroduction of appropriate substrate may be enough to stimulate the formation of new shellfish reefs (Bishop, unpublished data). The widespread commercial cultivation of *S. glomerata*, *C. gigas* and *M. galloprovincialis* is indicative that environmental conditions are suitable for shellfish growth and survival in many Australian estuaries.

Habitat requirements for shellfish reefs

Key environmental drivers of shellfish reef distribution include: (1) the availability of hard substrate; (2) water quality parameters such as temperature, salinity, and dissolved oxygen; (3) turbidity, and the availability of food resources; (4) tidal elevation (depth); (5) pollution; and (6) disease. Although all reef-forming shellfish require hard substrate, adequate food resources, and are sensitive to low dissolved oxygen and pollutants, they vary markedly in their salinity and temperature optima, their depth distributions and the diseases to which they are susceptible. Hence, a thorough assessment of site conditions should be done prior to selecting the species or appropriateness of shellfish reefs for risk reduction.

Hard substrate

The larvae of reef-forming bivalves require a hard substrate on which to settle. Historic overharvest of shellfish using destructive dredge methods, not only removed live shellfish, but also the dead bivalve shells that formed the foundation for reef establishment and growth[285]. Consequently, in many parts of Australia (and indeed the world), substrate is regarded as the key factor limiting shellfish reestablishment following cessation of destructive fishing practices.

Water temperature, salinity and dissolved oxygen

Temperature and salinity are two key environmental variables that affect the growth and survival of marine bivalves, and hence, their distribution[287, 288]. The three key native species of reef-forming shellfish in Australia each vary markedly in their thermal and salinity optima (Table 2.9). Within species, early life-history stages are particularly sensitive to these variables, displaying narrower ranges of tolerance[287, 288]. Many adult shellfish are robust to short-duration changes in salinity, closing their valves for days to weeks to avoid adverse conditions[289].

When evaluating whether an environment will support a particular reef-forming bivalve, the extremes (i.e., maxima and minima) as well as the average values of temperature and salinity at a site should be considered. Particularly within estuaries, these variables can display high spatial and temporal variation. Although, within estuaries, salinity generally declines with distance from the mouth, the precise location of freshwater inputs (either natural or human-made, such as waste-water discharges) can lead to anomalies in this pattern. Additionally, salinity in estuaries can display marked variation across tidal cycles (due to the migration of the salt wedge up and down estuaries) and across days or seasons (due to rainfall events that can lead to large freshwater pulses). In evaluating the thermal environment, it should be kept in mind that intertidal oysters will not only be affected by water temperatures, but also air temperatures during aerial exposure at low tide, the latter of which are often more extreme[290]. In general, water temperatures display greater diurnal and seasonal patterns of variation in shallow estuaries (where they are greatly affected by air temperatures) than in the coastal zone, where the thermal mass of seawater buffers variation.

As well as directly influencing shellfish survival, temperature and salinity can also indirectly influence survival by determining the susceptibility of shellfish to some diseases[291, 292]. This appears to be mediated, at least in part, by the weakening of bivalve immune systems under stressful conditions[293]. For example, in the Hawkesbury River, infection of Sydney rock oysters with the QX disease-causing agent, *Martielia sydneyi,* does not appear to occur at temperatures below 21.5°C and can be triggered by freshwater events[294], when salinity drops as low as 10 PSU and weakens the oyster's immune system[293]. The temperature dependence of this disease is consistent with greater susceptibility of more northerly distributed populations of the Sydney rock oyster to the QX-causing parasite, despite the distribution of the parasite as far south as the Victorian border[295].

Table 2.9. Temperature and salinity tolerances, and depth ranges of reef-building Australian native oysters.

VARIABLE		S. GLOMERATA	S. CUCULLATA	O. ANGASI
Water temperature (°C)	Larvae	19 – 28	20 – 35	20 –29
	Adult	11 – 36	5 – 40	8 – 29
Salinity (PSU)	Larvae	20 – 39	15 – 40	25 – 35
	Adult	15 – 55		21 – 46
Depth (m relative to MLW)		-8.0 – 0.5		-40 – 0

Chl *a* and turbidity

Oysters and mussels are filter feeders, consuming phytoplankton and detritus that pass through their gills. Hence, the concentration of Chl *a* (proxy for phytoplankton biomass) in waters can be a controlling factor in bivalve growth[296, 297]. Under natural conditions, growth may be limited by a lack of food, particularly near the mouths of estuaries where coastal conditions of low productivity may prevail. In oligotrophic eastern Australian estuaries, moderate anthropogenic nutrient loading of estuaries, can enhance Chl *a* and detrital concentrations, resulting in enhanced bivalve growth[298].

Whether turbidity has positive or negative effects on bivalve growth and survivorship depends on the concentration of suspended materials in the water, as well as its quality. Whereas high amounts of sediment resuspension can inhibit feeding, through the inhibitory and dilution effects of high inorganic particle loads, moderate amounts can enhance feeding by providing a supplementary food source[299, 300]. In oligotrophic waters where phytoplankton concentrations are seasonally low, resuspended benthic organic matter can serve as the primary food resource for bivalves in months of low phytoplankton productivity[301, 302]. In environments with high sediment loads, shellfish can be smothered by sediment deposition.

Tidal elevation

The reef-forming shellfish common in Australian waters vary in their depth distributions (Table 2.9). All shellfish must be underwater for at least part of the tidal cycle in order to feed, but species vary in their capacity to withstand the high temperatures and desiccation stress that occurs in the intertidal zone at low tide[303]. *Saccostrea glomerata* generally displays a greater tolerance to high temperatures and desiccation stress than *C. gigas* or *O. angasi,* allowing it to extend higher into the intertidal zone[303]. In general, shellfish growth rates decline with the proportion of the tidal cycle they are out of water, due to diminished feeding opportunity[304]. For *S. glomarata*, survival rates, conversely, may initially increase with elevation. This is because aerial exposure can help to control those parasites and competitors of shellfish that are sensitive to drying at low tide[304], and may reduce the time shellfish are exposed to finfish predators[305]. Nevertheless, in the high intertidal zone, the constraints of the narrow window of inundation across which feeding can occur, and the high temperature and desiccation stress eventually limit survival, with the maximum height of the reef crest set at approximately mean high water.

Pollution

Through their suspension feeding, bivalves can accumulate contaminants at concentrations greater than the surrounding environment[306]. Bivalves are particularly sensitive to tributyltin oxide (TBTO), the active ingredient used in Tributyltin antifouling paints. In laboratory studies, growth of both the Sydney rock oyster and Pacific oyster was reduced by up to 50% by as little as 5 ng TBTO[307]. High TBTO concentrations affected Sydney rock oyster industries in major NSW oyster growing estuaries (e.g., Hawkesbury and Georges River) during the 1980s[306]. Subsequently, the use of antifouling paints containing TBTO has been domestically and internationally banned, and the prevalence of shell deformities in oysters has declined and growth has improved.

Besides directly impacting shellfish, pollutants may also indirectly affect their survival by suppressing the bivalve immune system, rendering the animals more susceptible to diseases[308]. Disease-causing parasites can exist sub-lethally within shellfish populations, with stressful conditions triggering mortality events.

Disease

Little is known about the effects of disease on wild shellfish populations of Australia. Knowledge of disease primarily comes from the aquaculture industry where viruses, bacteria, protozoans and multicellular parasites or pests such as mudworm and flatworms have been documented to affect production, in some instances producing complete loss of stock[292]. These disease-causing agents may be host-specific or general.

The main diseases of *S. glomerata* are QX disease, caused by the protistan parasite *M. sydneyi*[295], and winter mortality, which is still of unknown cause[309]. QX disease has been detected in wild oyster populations, though it appears it may be far less prevalent and cause significantly less mortality than in aquaculture populations[310]. Because outbreaks of QX disease are often associated with freshwater runoff events, QX disease is most likely to be problematic for upstream populations of *S. glomerata*. Winter mortality, by contrast, affects oysters at high-salinity sites[309].

Like other *Ostrea* spp., *O. angasi* is susceptible to *Bonamia* spp. parasites with *B. exitiosa* specifically identified as causing mortality[311]. Little is known about the distribution of *B. exitiosa* in Australia due to the small *O. angasi* aquaculture industry and scant remnant wild populations.

In general, subtidal populations of shellfish are more susceptible to disease than intertidal populations due to their greater exposure time to water-borne disease-agents and because parasites may desiccate during low-tidal aerial emersion in the intertidal zone. For example, intertidal cultivation of Sydney rock oysters controls the shell-boring mudworm, which can cause unsightly mud blisters and lower oyster condition. Intertidal oyster restoration is more relevant to coastal protection due to the greater effect of intertidal reefs on wave attenuation (see Section: Performance factors for hazard risk reduction, below).

Approaches for hazard risk reduction

In ascertaining whether the construction of shellfish reefs might be a viable option for hazard risk reduction at a given site, the first steps are to ascertain: (1) the environmental suitability of the site for supporting the shellfish reef both now and into the future; and (2) the feasibility of creating a shellfish reef of the dimensions that are required to dissipate wave energy and protect shorelines from flooding and erosion (see section on Performance Factors below for further information).

Habitat Suitability Modelling is a GIS-based tool which can assist in deciding which sites are environmentally suitable for establishment of shellfish reefs[312]. Habitat Suitability Models utilise species-habitat relations and geospatial environmental data to create a composite suitability index with values ranging from unsuitable to suitable. When combined with shoreline exposure data, these models can identify areas where establishment of shellfish reefs is most likely to stabilise shorelines[313]. The environmental data included in a Habitat Suitability Model for shellfish reefs should be guided by the above list of factors influencing shellfish reef establishment, as well as data availability. Exposure data might include fetch, wind direction and wind speed. Habitat suitability models can also be combined with estuarine use data (e.g. exclusion zones for aquaculture, navigation, recreation) to produce Restoration Suitability Models, that also take into consideration estuarine uses and extant habitats that may constrain areas that are acceptable for the establishment of a shellfish reef.

If shellfish reef construction is indeed a viable option (Table 2.10), the next step is then to determine whether the absence of reef is due to ‘recruitment limitation’ (the absence of a source of shellfish larvae), ‘substrate limitation’ (the absence of suitable hard substrate to facilitate reef growth), or both[314]. In some instances, nearby remnant reefs and/or shellfish farms will provide an adequate supply of larvae to support reef development, without the need to actively introduce bivalve stock. Recruitment limitation can be assessed by placing substrate (e.g. concrete pavers) at the site and assessing whether shellfish colonise[315]. Substrate should ideally be introduced to the location shortly prior to commencement of the peak spawning period and, ideally, left in place for at least 1 year to account for trickle spawning that may occur throughout the year. Pilot studies, in which adult bivalves are transplanted into sites slated for shellfish reef creation can be useful in confirming that environmental conditions are indeed suitable to support reef establishment.

Table 2.10. A summary of the interventions for risk reduction using shellfish reefs. × = no; ✓ = yes; ? = information not known. The costs given are minimum – maximum.

		SUBSTRATE PROVISION	SEEDING FROM AQUACULTURE	SEEDING FROM WILD STOCKS
LIMITATIONS TO ESTABLISHMENT	**Overcomes substrate limitation**	✓	×	×
	Overcomes propagule limitation	×	×	✓
	Effective against top-down drivers*	✓	×	×
	Effective against bottom-up drivers*	✓	×	×
	Addition of resilient genotypes	×	✓	✓
	Propagation of resilient genotypes	×	✓	×
APPLICATION	**Scalable to large areas (> 1 km^2)**	✓	×	×
	High technical expertise required	×	✓	×
	History of use (for restoration)	✓	✓	✓
	History of use (for risk reduction)	✓	×	×
	Time to effectiveness (yrs)	0 - 5	1 - 5	1 - 5
	Set up cost (AUD m^{-2})	< 1 - 432	300	?
	Maintenance required	×	✓	×

*top-down drivers include predators; bottom-up drivers include sedimentation

Substrate provision

Hard substrate may be introduced in a variety of forms, and should be selected according to (1) its suitability for shellfish settlement, (2) the height, size and shape of shellfish reef to be constructed, (3) material availability and (4) logistics[316]. If shellfish reef construction is utilising natural recruitment, or juvenile bivalve seed, the substrate needs to provide microhabitats that will protect juvenile shellfish from predators that could otherwise limit reef establishment. Particularly in muddy areas, the substrate should also stabilise the underlying sediment so that its resuspension does not smother shellfish. In environments with high sedimentation, substrates should be sloped to minimise sediment accumulation.

Bivalve shell is the natural substrate on which reefs form, so is often the preferred substrate option in shellfish reef construction[315]. Whether this is a viable option will, however, depend on an adequate shell resource of an appropriate type being available, and an effective method of shell stabilisation. Increasingly shell is being made available through table-to-reef recycling programs utilising shell debris from restaurants serving shellfish[317]. Oyster and scallop shells have been used successfully as a substrate for oyster reef restoration, though use of mussel shell has been unsuccessful, at least for *O. angasi* (B. Cleveland, pers. comm.). Prior to reintroduction into estuaries and bays, shell must first be appropriately treated to remove potential parasites, pathogens and other pest species that may be translocated with the shell (this typically involves aging for at least 6 months in the sun[315]. The shell then needs to be stabilised. Historically this was done using bags or gabion baskets (preferably constructed of natural or biodegradable materials rather than plastics), but in environments with high sedimentation these may capture sediment and increase shellfish smothering. An alternative approach is to complement shell with large boulders that provide stability and trap shell.

Where shell is not available or infeasible to use, crushed rock with a calcium carbonate component (e.g., limestone) may be an alternative substrate, that provides a large surface area for larval colonisation and the protective microhabitats (between rocks) needed for survival of juveniles. Crushed rock has the advantage that it does not need to be bagged prior to deployment. Alternatively concrete units (e.g. blocks, oyster castles) may be used, as bivalves typically respond positively to the chemical cues released by concrete[318]. Additionally, a range of new biodegradable products (e.g. BESE-elements® starch mesh) are available, some of which provide refuge from predators[225].

Many of these substrates provide a wave attenuation and shoreline stabilisation function in their own right[69]. In selecting which substrate to use for reef establishment, a key consideration should therefore also be the proportion of shoreline stabilisation to be provided by the shellfish themselves versus the underlying structure[69].


Left: Oyster shell contained in coir bags, NSW © OceanWatch. Right: Oyster shell and rock contained in steel cages, VIC © Ralph Roob.

Active restoration of bivalves

Where there is an inadequate larval supply to support shellfish reef development, bivalves may be introduced as either adults or juveniles (the latter, sometimes also referred to as spat or seed[315]). Adults are generally more resilient to environmental stressors, less susceptible to predation and have greater reproductive output, but juveniles are often preferred because they can often more readily be obtained in high numbers, at low expense. Both adult and juvenile bivalves can be sourced from shellfish farms or from the wild. In both instances, biosecurity protocols should be adhered to in moving shellfish between estuaries, so as not to inadvertently introduce pathogens, parasites or invasive species with the shellfish. Additionally, harvest of wild shellfish should only occur where it does not compromise wild populations and the ecosystems that they support.

To obtain juvenile oysters, substrate (e.g., shell) on to which larvae can settle can be placed in a high-recruitment area. Following settlement of larvae, the substrate is then transported to the site of reef establishment. Hatcheries may also settle shellfish larvae onto shell, or other larger pieces of substrate, to produce 'cultched' seed or, alternatively, onto sand grains or small shell fragments that quickly get eclipsed by the size of the growing animal ('cultchless' seed). The latter is commonly used by aquaculture industries that grow out shellfish in bags and racks, and that target the half-shell market (i.e., oysters served on the shell on a plate). Where available, however, cultched seed (i.e., shellfish settled onto shell) can be a better option for shellfish reef construction, because (1) it replicates the natural reef structure, where larvae attach to shells produced by previous generations, (2) the complexity of shell can help to protect juvenile bivalves from predators and other environmental stressors (e.g., through enhanced shading or moisture retention) and (3) this method achieves the dual purpose of introducing substrate and seed. Use of cultched seed may be particularly desirable in highly dynamic environments because the increased weight of the shell provides greater stability than the highly mobile cultchless seed that can be lost within the complex structure of larger substrates.

In some instances, logistics may constrain the source of seed for shellfish reef construction. However, in instances where multiple sources of seed are available, the principles of provenance and maximisation of genetic diversity can be helpful in identifying the most appropriate mix of seed. Historically, restoration practitioners have targeted locally sourced seed for transplant into restoration sites under the presumption it would be better adapted to the local environment, and preserve the local gene pool[319]. However, it is now recognised that the exclusive use of local material, may constrain rapid evolution to anthropogenic and climatic stressors[320]. Instead, predictive provenancing, where stock is acquired from sites matching projected future conditions, or admixture provenancing, where seed is mixed from many populations throughout the species' range, are being advocated as potentially more successful at restoring resilient reefs. These approaches form part of a move toward climate adaptation through assisted gene flow[321] and maximising the genetic variation on which evolution can act[322].

Increasingly, selectively bred genotypes targeting fast growth and disease resistance are available through aquaculture industries[323]. Studies with selectively bred Sydney rock oysters have shown that while these can display superior performance to wild-type genotypes in aquaculture settings, this is not necessarily the case in the more natural settings in which reefs are constructed[290]. This is because there are trade-offs between these selected traits and tolerance to other environmental stressors, such as high temperatures, that can render selectively-bred oysters more susceptible to extreme heat events[290]. Consequently, any proposal to use a single line of selectively bred shellfish for reef construction should proceed with caution. In order to maximise resilience to a range of stressors, a better strategy might be to maximise genetic diversity rather than select for certain genotypes.

Performance factors for hazard risk reduction

Reef dimensions

Shellfish reefs have often been designed using principles from low crested breakwaters, where key factors that can influence hazard reduction performance include the porosity and roughness of the structure, reef crest height and width, and freeboard (i.e., difference between structure height and still water depth)[324, 325]. The majority of our knowledge on the wave attenuation of shellfish reefs is based on observations of living shorelines using oyster reefs in the United States, which tend to be narrow, fringing reefs with the purpose of erosion control. As there are not many reefs that remain[285], observations of wave attenuation under more natural conditions are scarce. Shellfish reefs are most effective at reducing the wave-driven contribution to coastal erosion and flooding (see Section 1.2). Like low-crested breakwaters, shellfish reefs produce greatest wave attenuation when the crest of the structure is at or above the still water level[281, 282]. However, reef crests that spend a greater proportion of their time exposed, while maximising wave attenuation are not suitable habitat for oysters due to the amount of aerial exposure[69]. The height of the structure relative to the water level is a function of the absolute elevation of the reef and the tidal range. Reefs at locations that have large (macro) tides will have more variation in the depth of water above the reef crest compared to those with smaller (micro) tides. As wave attenuation also increases with crest width, which enhances drag dissipation due to the porosity and roughness of the reef (see Section 1.2) a wider reef may partially compensate for a lower crest height[282]. As shellfish reefs are physiologically constrained in the maximum tidal elevation they can persist at, wide shellfish reefs may be more feasible structures for wave attenuation.

Reef condition

The presence of oysters is a necessity to gain the sustained and adaptive advantages of nature-based methods over traditional structures. However, oysters also increase bed roughness, and therefore drag, resulting in greater energy attenuation[326, 327]. A greater coverage of oysters, as well as closed reefs (i.e., few open patches) results in greater control on erosion and flooding[328]. Loose shell can be moved around in wave events, which is why, in greater energy environments, shell substrate should be contained to hold it in place while shellfish recruit, and it is important to understand if reefs are recruitment limited (as discussed above). If not colonised by oysters, reefs could eventually disintegrate[329].

Coral Reefs

Coral reefs are three-dimensional calcium carbonate ($CaCO_3$) structures successively built from generations of reef skeletons. Reef-building corals (Phylum Cnidaria, Class Scleractinia) live in symbiotic association with the algae Symbiodinium, which provide the energy that fuels coral growth and calcification[330]. When alive, corals provide a structurally complex habitat on the reef's surface that supports high biological diversity[331], and on death the coral skeleton contributes to long-term reef accretion[332]. Corals are morphologically complex, taking a range of forms such as branching, foliose and massive. Reefs with higher coral cover and morphological diversity are more structurally complex, which equates to greater habitat provision for reef organisms[331]. Reef structural complexity, together with reef depth, is also critical for shoreline protection, as the greater the complexity or roughness of the reef surface, the higher the influence on wave energy dissipation and transfer over the reef[333, 334]. High reef roughness (also termed reef rugosity) therefore has a significant dampening effect on wave action and can reduce coastal erosion[335]. As well as acting as natural breakwaters, coral reefs provide carbonate sediments that can support and maintain beaches along reef fronted shorelines, which also act as a buffer between waves and land[336].

Branching, foliose and massive forms of coral. Eva Reef, Queensland © Nicola Browne

Coral reefs largely occur within warm (21 to 29 °C), nutrient-limited, clear waters where conditions are traditionally considered optimal for coral growth[337]. Extensive coral reef ecosystems line most of Australia's northern shores. These include the Great Barrier Reef (GBR) system along the east coast that extends southwards from the Torres Strait in the north to Frazer Island, and the numerous reefs along the north and west coasts that extend from the western coast of the Gulf of Carpentaria westwards towards the Kimberley reefs and southwards towards Ningaloo on Western Australia's central coast and the offshore Houtman Albrolhos islands (Figure 2.1). Coral reefs of the GBR are the most well studied of Australia's reef systems, although there has been a significant increase in the number and diversity of studies along the Western Australian coastline in the last 30 years[338].

Australia's coral reef systems include a variety of reef types that have arisen in response to climatic and tectonic processes, including nearshore fringing and patch reefs, offshore atolls and extensive carbonate barrier reefs[339, 340, 341]. Reefs can be further classified by their environmental drivers, which dictate community composition and reef growth. For example, reefs can be classified as tropical to temperate depending on mean sea surface temperatures[342], and clear-water to turbid depending on in-water light conditions and sediment loads[343]. Reefs growing in temperate regions (at high latitudes) and in turbid waters are often considered to be marginal reefs and/or non-reef building systems where conditions for coral and reef growth are sub-optimal[344]. While there is growing evidence that these marginal reefs and reef communities can have high coral cover and support diverse and highly adaptable corals, non-reef building coral systems are less relevant for coastal protection[345, 346]. In Australia there are both turbid reefs (e.g., inshore GBR, Kimberley, Exmouth Gulf) and high latitude temperate to semi-temperate reefs and reef communities (e.g., Rottnest Island, Lord Howe Island). These reefs are at the environmental limits of growth, and, as such, will be useful indicators of ecosystem shifts (e.g. range extensions) with future climate change. Anticipating these ecosystem shifts will be critical in evaluating the ability of coral reefs to continue to provide services, such as coastal protection, as well as identifying sites for coral reef establishment.

Habitat requirements for coral reefs

Key environmental drivers of coral reef distribution, composition and growth include temperature, aragonite saturation, water characteristics (e.g., light, nutrient and sediment loads), ocean currents and the history of sea-level change[347, 348]. Appropriate site selection is key for coral reefs as many of their habitat requirements relate to large-scale processes that are difficult to manipulate (Table 2.11).

Water temperature and salinity

Water temperature can vary over small (metres) to regional scales (100 km's) and temporally over months and seasons. Temperature has a significant influence on calcification rates and, therefore, faster rates of coral growth (and reef accretion) are typically located in warm waters (21 to 29˚C). Corals can exist outside these ranges but will transition from 'true coral reefs' to 'coral communities'[349], typically characterised by lower reef accretion rates and, therefore, a lower capacity to keep up with rising sea-level. Localised site characteristics heavily influences temperatures on coral reefs. These include basin characteristics such as the depth and shape of the basin. For example, semi-closed systems (e.g. in semi-enclosed embayments) have longer water residency times resulting in warmer temperatures compared to more open systems with faster flushing rates and shorter residency times[350]. Thus, localised site characteristics are important when assessing the future stability of a coral reef with rising sea surface temperatures and sea, as well as assessing a site for coral reef construction.

Salinity within coastal systems also varies over small to regional scales and influences rates of calcification. The optimum range that promotes faster rates of calcification (and therefore reef accretion) is from 32 ppt to 40 ppt[337]. It is influenced by precipitation, evaporation and riverine freshwater delivery. In Australia, coral reefs typically experience lower salinity during the wet summer months due to more rainfall[351]. Coral reefs situated close to river mouths (<50 km depending on size of the river) will also experience more variable salinity (as well as nutrients and light) than those further away[352]. Reefs that have initiated and grown close to river mouths will have adapted to the local environmental conditions (e.g. Kimberley reefs) but could be considered as marginal reefs at the edge of their environmental tolerances. These reefs may, therefore, be more vulnerable to future climate change and require additional support (e.g. improved water quality) to conserve reef functions (e.g. coral growth) and related ecosystem services (e.g. rugosity and coastal protection). For assessing site suitability for a coral reef construction project, knowledge on seasonal changes and drivers of salinity (together with other water quality parameters discussed below) will be a key determinant of project success.

Table 2.14. Habitat requirements for coral reef development.

VARIABLE	OPTIMUM	FULL RANGE	THRESHOLD
Water temperature (°C)	21-29	16-34	
Salinity (ppt)	32-38	25-42	
Nitrates (µg L^{-1})			<2 to 8
Phosphates (µg L^{-1})			<5
Surface light (µE.m^{-2}.s^{-1})			>450
Benthic light			50 to 450
Turbidity (mg.L^{-1})			10
Sedimentation (g.m^{-2}.day^{-1})			10
Aragonite saturation (Ω)	~3.83		>3.5

Waves and currents

Environmental parameters that fluctuate spatially across 10's of kilometres and temporally over days include physical parameters such as waves and currents. These local-level habitat requirements are also interrelated with water quality parameters, such as nutrients and pollutants, as water movement from waves and currents influences the delivery and dispersion of these from rivers and land run-off.

While tides are an important source of water motion on many reefs, for wave exposed reefs the breaking action of waves also drives mean currents. Waves and currents influence coral productivity and growth[353], transport sediments[354], deliver nutrients and remove metabolic wastes[355], and are therefore important factors to consider for both maintaining reef function and establishing new sites for coral reef growth. For example, sites with lower waves that have lower circulation rates and higher residence times (e.g. > 2 weeks), are potentially more susceptible to anoxic conditions and coral mortality events[356]. Conversely, sites exposed to high energy events, are at greater risk from coral dislodgement, coral mortality and reef morphological changes[357]. As such, coral reefs situated in higher wave energy (more exposed sites) and/or high current locations are often dominated by coral morphologies less prone to breakage e.g. massive corals[358]. Currently, there are no defined upper wave and current energy thresholds considered detrimental to coral growth and reef development as a reefs response to increasing wave energy will depend on the size and skeletal density of the corals, the dominant coral morphology, the stability of the substrate, reef depth and rugosity. However, site selection for coral reef construction should consider both adequate water flow and circulation, which can be measured in situ, as well as the susceptibility of the coral reef to high energy events, which can be based on coral reef characteristics.

Nutrients and pollutants

Nutrients and pollutants from land-based sources including agriculture, deforestation and urban coastal development influence coral reef function. Important nutrients include nitrates and phosphates, and common pollutants include pesticides, oils and toxins. Elevated nutrient levels cause increases in macroalgal cover, which compete with corals for space on the reef[359], and can also cause phytoplankton blooms that reduce light availability for corals[360]. High nutrient levels can also be toxic to corals and reduce coral fertility[361]. The impacts of pollutants such as trace metals, pesticides and surfactants are diverse, and have been linked to increased rates of coral disease and mortality[362] resulting in reef degradation. As such, waters surrounding coral reefs, both new and established, need to be low in nutrients and pollutants[363].

Light and turbidity

The environmental parameters that typically fluctuate over fine spatial (metres) and temporal scales (minutes to days) include light and sediment dynamics (sedimentation and turbidity). Light is a critical requirement for coral growth and typically limits reef development at levels <450 µE m^{-2} s^{-1} at the benthos[364]. Light is primarily limited through attenuation with depth[364], although light attenuation can be significantly increased by suspended particulate matter (e.g. sediments[365]), which in turn is influenced by water flow. Sediments have a number of impacts on reefs that include reduction in light and abrasion when suspended, to smothering benthic organisms resulting in tissue mortality when deposited[366, 367]. Sediments can, however, also be a source of nutrition for corals[368], providing an additional energy source under low light conditions. It has been suggested that healthy coral reef systems require suspended sediments concentrations of <10 mg L^{-1} and sedimentation levels should not exceed 10 g cm^{-2} day^{-1} [369]. However, given the highly dynamic nature (over space and time) of sediment delivery and resuspension by waves and currents over reefs, it is important to acquire high resolution spatial and temporal data to provide an accurate assessment of sediment (and light) conditions on an established reef as well as at potential sites for coral reef construction.

Aragonite saturation

Aragonite saturation varies at both the regional and habitat level. In Australia, aragonite saturation decreases southwards along the coasts from 3.9 in the tropics to 3.3 in temperate regions[370]. A value of 3.5 is considered to be the transitioning threshold from healthy to marginal coral reef conditions[371]. When aragonite saturation levels fall below this threshold, the ability for corals to create new carbonate declines[372]. Aragonite saturation states can also vary considerably within a reef. For example, De Carlo *et al*[373] observed fluctuations in saturations states from <3 to >5 in a 24 hour period on a Dongsha Atoll reef flat in north Pacific, with minima recorded just before dawn. This illustrates that the biological process of calcium carbonate production by calcifying organisms can have a rapid and significant impact on water chemistry and needs to be considered when evaluating current aragonite saturation states on reefs. However, both reef restoration and site selection for establishing new reef structures need to evaluate the broader regional variations in saturation states and pay particular attention to future declines due to climate change. Further, for those coral reefs situated in regions where the current aragonite saturation state is close to the threshold, the risk of lower carbonate production rates and possible reductions in sediment supply needs to be evaluated in the context of the ecosystem services that these reefs provide.

Approaches for hazard risk reduction

Reef conservation measures include addressing the cause of reef degradation (assisted establishment), improving substrate quality and quantity (substrate stabilisation, addition and enhancement) or actively restoring the coral community (active restoration). These measures have traditionally been used to address declining reef ecological function and related ecosystem services such as biodiversity, fisheries and tourism, but are increasingly seen as effective measures that also reduce the risk of coastal hazards. Here we discuss these approaches and how they relate to coastal hazard risk reduction. Further, we summarise the requirements for each intervention (e.g. costs, resources and expertise required) with the potential advantages for each technique to aid decisions around which intervention is most appropriate for a site (Table 2.12).

Assisted establishment

Where coral reefs have been degraded due to anthropogenic stressors, a reversal of these stressors may be all that is required to restore a functioning system. This can be done through passive or assisted management approaches that largely involve the removal or reduction of threats to coral reefs through: 1) reducing the exploitation of reef resources (e.g., no take areas), 2) adaptive legislation, stakeholder involvement, education and enforcement (e.g. catchment management[374], and 3) the removal of predators (e.g., pest management) or competitors. Marine no take areas can provide indirect benefits for coastal hazard risk reduction through habitat preservation and the protection of ecological interactions that support ecosystem function; for example, by increasing herbivorous fish abundance that reduces macroalgal competition with coral. The goal of catchment management is to improve water quality in coastal areas where coral reefs are located, as poor water quality has well-documented effects on reef health and resilience to global stressors associated with climate change[375]. The removal of predators is considered an effective strategy, particularly in the Indo-Pacific region where a main contributor to coral-cover loss has been the Crown of Thorns Starfish (COTS[376]). Responses to outbreaks have involved the manual removal of individuals, cutting individuals into small pieces and injecting them with toxic chemicals[377]. In addition, pest management control measures include catchment management programmes to prevent future outbreaks[378] and the protection of natural predators of COTS by limiting fishing activities[379]. A main competitor with corals for space on the reef is macroalgae, which can negatively affect coral recruitment[380], growth and fitness[381]. Macroalgae can be manually removed, however, this is generally only useful at small spatial scales (< 100 m) where a reef has undergone a phase shift from a coral to macroalgal dominance[382].

Substratum stabilisation, addition and enhancement

Coral larvae need a stable hard substratum on which to recruit. An unstable substrate will impede reef recovery as it directly impacts the survival of coral recruits[383]. Substrate stabilisation typically involves the use of mesh or netting fixed over loose rubble to reduce movement of the substrate[384], although large boulders have also been used to provide a new stable substrate[385]. This technique is more successful on reefs where unstable substrate is the main issue preventing reef recovery and where larval supply is not an issue [385]. Substrate stabilisation is a low cost method and is often employed in tandem with other active restoration techniques such as coral transplanting[384].

If hard substrate is not present, then this can be artificially constructed, for example using complex concrete units as an artificial reef. This approach has, however, been rarely used for coastal protection in the past resulting in uncertainties on how to design and deploy structures for this purpose. Reguero *et al*.[386] reports on one of the rare reef restoration projects to date that was designed and engineered to provide coastal protection benefits. This project involved the use of historical and modelling data, geotechnical surveys of the seabed, and field testing and monitoring to provide adequate and sustainable hydrodynamic performance. Equally, the artificial reef substrate must also be designed to facilitate a functioning coral reef ecosystem, rather than expecting coral to recruit to a structure akin to a traditional breakwater. The more rugose the substrate at the micro (mm) and macro- scale (m), the greater the surface area for colonisation. Rugose substrates also provide sheltered areas that can protect recruits from high wave energy as well as predation. The fine-scale role of the substrate micro-structure is particularly important for coral recruitment and subsequent carbonate production on the reef[387]. Further, the orientation of the substrate (horizontal versus vertical) can greatly influence settlement and recruitment rates, with horizontal surfaces exposed to more light but also more sediments[388]. The offset between more light and greater sedimentation will likely have varying impacts on recruiting organisms due to environmental differences between sites and different species thresholds. When placing an artificial substrate, the supply of coral recruits, which is a key driver of reef recovery following disturbance events[389, 390], and the likelihood of coral predators are also important considerations. Given this approach has not been widely used, research will be needed to support the design of structures that achieve both engineering and ecological goals.

Table 2.12. A summary of the interventions for risk reduction using coral reefs. × = no; ✓ = yes; ? = information not known. The costs given mean – maximum.

		NO TAKE AREAS	CATCHMENT MANAGEMENT	PEST MANAGEMENT	SUBSTRATUM STABILISATION	SUBSTRATUM ADDITION	SUBSTRATUM ENHANCEMENT	TRANSPLATING CORAL FRAGMENTS	LARVAL ENHANCEMENT	ASSISSTED GENE FLOW TECHNOLOGY	ASSISTED EVOLUTION &/OR SYNTHETIC BIOLOGY
LIMITATIONS TO ESTABLISHMENT	**Overcomes substrate limitation**	×	×	×	✓	✓	✓	×	×	×	×
	Overcomes propagule limitation	×	×	×	×	×	×	✓	✓	✓	✓
	Effective against top-down drivers*	✓	×	✓	×	×	×	×	×	×	×
	Effective against bottom-up drivers*	×	✓	×	×	×	✓	×	×	×	×
	Addition of resilient genotypes	×	×	×	×	×	×	✓	✓	✓	✓
	Propagation of resilient genotypes	×	×	×	×	×	×	×	×	×	✓
APPLICATION	**Scalable to large areas (> 1 km^2)**	✓	✓	✓	✓	✓	×	×	✓	✓	✓
	High technical expertise required	✓	✓	×	×	✓	×	×	✓	✓	✓
	History of use (for restoration)	✓	✓	✓	✓	✓	✓	✓	✓	×	×
	History of use (for risk reduction)	×	×	×	×	✓	×	×	×	×	×
	Time to effectiveness (yrs)	5 - 10	5 - 10	5 - 10	2 - 5	<1	2 - 5	1 - 5	5 - 10	5 - 10	5 - 10
	Set up cost (AUD m^{-2})	0.001 - 40	?	24†	52-3650	460 - 20000	?	31-1200	72-600	?	?
	Maintenance required	✓	✓	✓	×	×	✓	×	×	×	×

*top-down drivers include predators; bottom-up drivers include water quality and competition with macroalgae; †number is per starfish

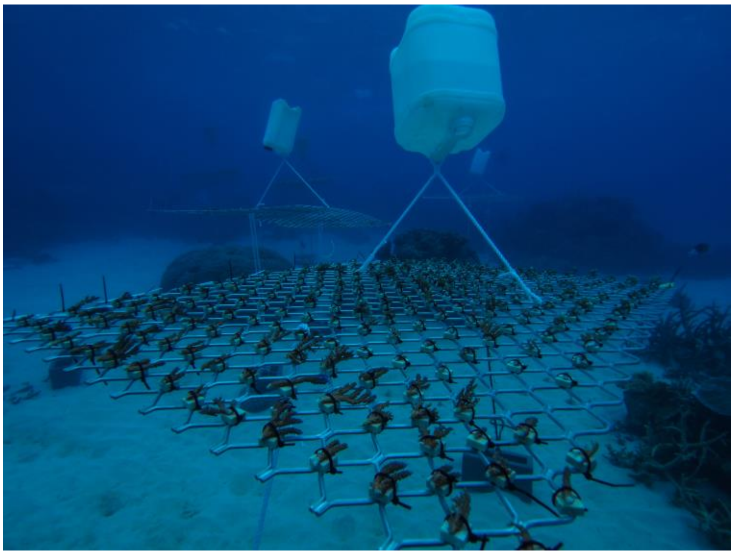
An artificial reef can incorporate substratum enhancement, where an electrical current is applied to an artificial reef structure with the purpose of increasing calcification rates of the coral polyps[391]. Bostrum-Eniersson *et al.*[384] reviewed data from nine studies that had applied the technique and found contradicting results with some studies reporting an increase in coral growth[392] and others a decline in coral growth[393]. These mixed reports may, in part, be due to different test coral species, but could also reflect the influence of other environmental drivers.

Active restoration of corals

If there is not a source of natural recruits, then the aforementioned techniques (with the removal of stressors as a critical first step; see Assisted establishment above) may need to be combined with active restoration of corals that involve transplantation of individuals or larval seeding.

Transplanting coral fragments

Coral cover (and reef structural complexity) can be restored by transplanting coral fragments on to the reef substrate. The most common method is direct transplantation, which uses coral fragments broken off from donor corals, and typically involves fast growing corals (e.g., *Acropora* spp.). This technique utilises the coral's asexual propagation capacity and, therefore, leads to the generation of clones[394]. Damage to donor corals and populations had led to a more sustainable approach known as coral gardening whereby coral recruits or small fragments are grown in nurseries. These nurseries protect corals from adverse conditions and fragments can grow to a suitable size before either outplanting on the degraded reef or further fragmentation to increase the number of fragments available [384]. The main advantage of both these techniques is that fragments have a higher chance of survival than coral recruits and they require less advanced expertise. The main disadvantage is that genetic diversity is low[395].



A coral nursery prior to outplanting © Emma Camp.

Larval enhancement (seeding) and assisted larval dispersal

Larval enhancement (or seeding) is based on the sexual reproduction of corals, which promotes genetic diversity. Large amounts of eggs and sperm are collected in the field and brought back to the laboratory for fertilisation. The embryos and larvae are reared in holding tanks before either being settled ex situ on to a range of artificial structures, which are then deployed on to the reef[396], or directly released on to the reef in enclosures that retain them to the specific site[397]. This approach has been shown to significantly enhance recruitment and re-establish breeding populations[397] and can up-scale restoration efforts beyond the average 100 m^2 (associated with coral transplantation). Further, establishing small but genetically diverse coral populations that will then reproduce sexually in situ, will enhance rates of reef recovery[398]. The key disadvantages are that it requires a high level of expertise, costly facilities for rearing larvae and takes longer for tangible results (e.g., increased coral cover) to be observed on the reef [395].

Assisted larval dispersal involves gene-flow technologies that may enhance the distribution of resilient genotypes[399]. This emerging technology is largely focused at facilitating the spread of genotypes that may enhance resilience to climate change (e.g., heat tolerance), and could work in conjunction with larval enhancement techniques. The enhancement of climate resilient traits can also potentially be achieved through assisted evolution and synthetic biology, although these technologies need more stringent procedures and checks[400].

Performance factors for hazard risk reduction

The development of performance factors or indices that link coral reef condition to coastal protection can be challenging due to the complexity of coral reef systems and the lack of suitable historical data across many parts of Australia (and the world more generally)[386]. Nevertheless, many of the underlying processes that determine the effectiveness of reefs to reduce coastal flooding and erosion risk are well-understood and can be forecast with modern predictive models[401]. Of those studies that have demonstrated a link between changes to coral reef condition and shoreline stability[402, 403], factors such as reef degradation and coastal development were considered to be important, but causality can still not be definitive[404].

Reef geomorphological characteristics

Coral reefs are efficient at dissipating wind wave energy over their shallow bathymetry (wave attenuation by depth-induced breaking) and the large roughness of reef organisms (wave attenuation due to roughness; see Section 1.2)[334, 405, 406]. Three important factors for wave transmission over the reef are reef width, reef depth (relative to wave height) and reef roughness. Reefs with high rates of wave attenuation are at least twice as wide as the wave-length and at depths less than half the incoming wave height[407]. Furthermore, reefs with more complex high-relief structures cause greater rates of wave attenuation through drag dissipation[278, 408]. Live coral provides much of the geometrical complexity on the reef, and hence loss of live coral cover will eventually result (i.e. following framework degradation) in an increase in wave energy reaching shorelines[335]. Yet reef wave attenuation studies rarely report on live coral cover (or species composition) or measure the physical properties of the reef roughness (i.e. rugosity[404]). In addition, field studies that have measured waves over reefs are typically carried out under ‘normal’ wave conditions. Therefore, there have been limited field studies of extreme wave conditions with reef geomorphological characteristics[409]; however, a number of physical modelling studies using scaled models of reefs in large-scale experimental facilities have provided some detailed insight[401].

While wave breaking over reefs attenuates incident wind waves that contribute to wave runup and coastal erosion, the wave forces generated during breaking can also generate other forms of water motion that can sometimes reduce the coastal protection function of reefs (see Section 1.2). This potentially includes a rise in the mean water level (wave setup) shoreward of reefs[410] and the generation of low-frequency (infragravity) waves[411], which together can become the major cause of coastal flooding behind reefs[401]. Similarly, wave breaking on reefs can generate mean currents (much like rip currents on beaches), which can either lead to beach accretion or erosion depending on the predominant flow patterns that occur[412]. These other (often more complicated) hydrodynamic processes that occur on reefs must be carefully considered when designing reefs for coastal protection, which depends on how a reef geometry interacts with the incident wave conditions.

Reef condition

A healthy coral reef will have shorter (< 10 years) recovery times following disturbance events (e.g., storms and cyclones). Reefs that are able to recover from disturbance events will regain associated ecosystem services (e.g., coastal protection) at a quicker rate. Hence, reef characteristics that promote reef recovery, such as coral cover, diversity, proportion of breeding corals and larval supply, are potentially useful performance indicators related to risk reduction. These characteristics are typically monitored for reef conservation projects[384], so related protocols could easily be applied for assessing long-term coastal protection benefits.

A comprehensive method that assesses reef condition, stability and growth is the carbonate budget method. Carbonate budgets quantify net carbonate production through measuring all sources of carbonate production (e.g., corals) and carbonate removal (e.g. bioerosion[413, 414]. Reefs with a net positive carbonate budget are stable and/or accreting. These reefs could potentially keep up with sea level rise thereby maintaining their coastal protection services. Conversely, reefs with a negative carbonate budget are eroding and will likely lose their ability to protect the shoreline [415, 416]. Hence, the reef carbonate state could be a useful indicator for assessing reef performance with sea level rise projections.

Waves breaking over the Great Barrier Reef © Ian Young.

Box 2.2 Multi-habitat Restoration

Most nature-based risk reduction projects to date have focused on the recovery of one habitat. However, in natural systems, mosaics of ecosystems are often observed, for example those that span a tidal or depth gradient. The restoration of one habitat may increase the resilience of another, for example oyster reef living shorelines are often deployed adjacent to saltmarsh to reduce erosion in the United States. Recovery of one habitat can incidentally lead to the establishment of another. For example, in the lee of a shellfish living breakwater installed in Victoria (see Box 3.1) dense seagrass patches have established, and the accumulation of sediment on the shore has created a small berm, which has been planted with dune vegetation. Planting of coastal trees such as *Casuarina* is often integrated with the installation of mangrove-rock fillets (see Box 2.2) to further stabilise estuary banks in NSW. A combination of habitats can increase the energy reduction that occurs with one habitat alone[327]. Further, many species require different habitats at different life-history stages, so the use of multi-habitat restoration has the potential to support greater ecological outcomes, and therefore maximise co-benefits[417]. Diverse systems can also have greater resilience to climatic disturbances (see Section: Designing for the Future)[418]. One negative of risk reduction approaches that employ multiple habitats is that more space will be needed to account for the different environmental requirements for each species used. However, if the approach combines an offshore and onshore method, the terrestrial space requirements will be reduced (e.g., shellfish reef with dune management).

Designing for the Future

It is common practice to design traditional infrastructures for future conditions, accounting for potential sea level rise or storm events that may occur within the planning timeframe (refer to NCCOE, 2012[419]). One of the potential advantages of nature-based methods is that they are adaptive to changes in climate over longer timescales than traditional structures (see Section: Benefits of Nature-Based Methods). This may mean that nature-based methods do not lose efficacy at providing risk reduction, for example by maintaining height relative to sea level through accretion or growth[420]. However, marine and coastal environments are rapidly changing and predicted to be subject to an increasing number of multiple stressors, which can impact both natural and restored ecosystems. Future-proofing[421], therefore, needs to be considered and built into the design of nature-based methods. Examples of methods for supporting resilient populations include preserving genetic diversity, which may involve introducing genotypes of already adapted species (e.g., to warmer conditions; 'assisted gene flow') or those that have been artificially selected for through breeding under future conditions ('assisted evolution')[400]. Given the importance of maintaining the risk reduction service, nature-based methods may focus on restoring functionality, rather than a target species, although this could result in increasingly novel ecosystems[421]. Designing for the future requires determination of what period of time we are restoring for by choosing an appropriate planning horizon. For longer timescales, adaption may need to be done gradually, for example by introducing resistant or adapted genotypes over time as the conditions change[422]. Importantly, maintaining space for natural dynamics, which includes shoreward movement, if needed, will support increased resilience of nature-based systems and the protective services they provide.

3 Implementation Framework

The implementation framework is based on the four project development phases that are applied to any infrastructure project. In this section we highlight considerations that are specific to nature-based methods during the planning, assessment, design, construction, monitoring and maintenance steps of a project (Figure 3.1). We do not provide detail on the steps common to nature-based and traditional methods, and this framework should be read in parallel with the National Committee on Coastal and Ocean Engineering Guidelines (NCCOE) for climate change and coastal adaptation[419]. This section brings together existing frameworks for nature-based methods by Ecoshape[423], PIANC[424], and The World Bank[425].

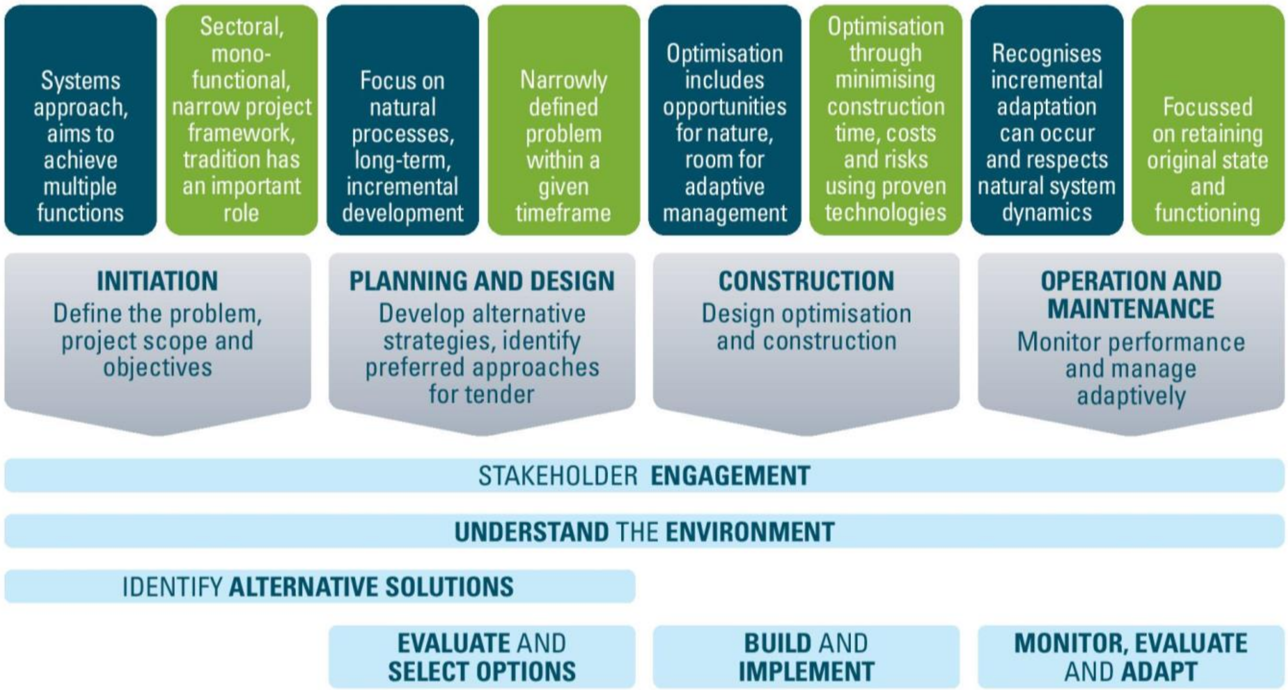


Figure 3.1: The four phases and steps (light blue boxes) for implementing a project using nature-based methods. The focus of nature-based (dark blue boxes) and traditional (green boxes) methods during each phase is highlighted.

Step 1: Initiation

The initiation phase that includes nature-based methods differs from that focussed on traditional structures in that the aim is to achieve a structure that provides multiple benefits (e.g., to nature, for recreation), as well as fulfilling a risk reduction purpose. In contrast, the initiation of a hard structure takes a mono-functional approach, using a narrow project framework that is based on tradition and is led by a single sector or authority.

Given the multi-functional aspect of nature-based methods, early expertise is required from other disciplines that may include ecologists, geomorphologists, social scientists, and local Traditional Owners, in addition to engineers, with full engagement of all relevant stakeholders. Previous experience suggests that having a mix of people that are experts in the traditional solutions and those that have experience working with natural systems and nature-based methods produce the best results in terms of innovation and feasability[423].

Define the study area, problem, and key stakeholders

Identify the area of interest and the hazards that area is exposed to. Determine the boundaries of the natural and socio-economic systems within that area. Identify the key stakeholders associated with the intervention area and set-up an engagement process to understand their views and expectations[426].

Set the project objectives

Objectives are usually focused on the engineering functions of the infrastructure and ongoing operational and economic requirements. With nature-based methods, objectives must also be defined based on ecological function. Other ecosystem services, such as carbon sequestration, water filtration or social, cultural and recreational value may also be included as objectives to achieve the multi-functional aim of nature-based methods.

Step 2: Planning and Design

The planning and design phase involves developing alternative strategies within the scope and selecting the preferred approaches for the tender process. When using nature-based methods the focus is on the longer-term, with recognition that the solution will develop over time and can be adaptively managed. The focus of a traditional structure is shorter term, with the aim to solve a narrowly focused problem within a given timeframe.

Understand the environment

A detailed hazard, ecosystem, and risk assessment is needed. A coastal process assessment that includes knowledge of the current and future hazards should be based on a review of the existing literature/datasets, field studies and modelling, the steps for which are described in the NCCOE guidelines[419]. Key to nature-based methods is an understanding of the ecological environment. This includes whether current ecosystems exist, and their extent, condition and functioning. Historical trends in current ecosystems can inform their stability and resilience, while likely future patterns of distribution need to be considered. Similar to coastal processes, these can be modelled using species distribution models. Ecosystem health can be measured using metrics such as species diversity, abundance and biomass. The role of existing ecosystems in reducing risk through regulating hazards (e.g., wave attenuation), reducing the exposure of people to hazards (e.g., by reducing occupation of the hazardous zones) and reducing vulnerability (e.g., by supporting livelihoods and other services) should be assessed[425]. Where ecosystems aren't present, an understanding of the environmental conditions important for their establishment is required to determine the feasibility for different nature-based methods (see Section 2 for habitat requirements). The risk is assessed based on the hazard, exposure (e.g., number of people, value of properties) and vulnerability (e.g., building types, demographics).

Identify alternative solutions

Nature-based methods can be considered within an adaptation pathway approach (see Section 1: Considerations for Nature-based Methods). The target for hazard reduction should be identified based on the acceptable level of risk, and in consultation with stakeholders. There is increasing data available on the wave attenuation and sediment stabilisation of natural and restored coastal habitats (see Section 2) that can inform the benefit of conserving or expanding an existing ecosystem, or the restoration of a historical ecosystem. The identification of alternative solutions should be based on the technical feasibility from an engineering and ecological perspective (see Section 2), the economic viability (see Section 4), the governance framework and regulations (see Section 5) and the social license to implement that solution. Stakeholders should be consulted on the potential intervention strategies and given they may be less familiar with nature-based methods, their role in risk reduction based on the risk and ecosystem assessments should be discussed. While traditional structures are often funded through local and state government infrastructure funds[43], nature-based methods may open up other avenues of funding, such as for environmental protection. Given the increased interest in upscaling marine habitat restoration in Australia[427], targeting areas that also provide coastal hazard risk reduction is another consideration.

Evaluate and select options

A full cost-benefit analysis and modelling of the alternative options within the risk model can be used to identify the most effective option. Methodologies and models for evaluating nature-based methods are under continual development, and learnings should be taken from existing projects. Given that nature-based methods can have a range of environmental and societal benefits, these should all be identified and included in the benefit-cost model (see Section 4). The most effective and appropriate action based on the problem, cost-benefit analysis, stakeholder views and capacity should be selected and designed.

Step 3: Construction

A large focus of nature-based methods is the optimisation of the cost-effectiveness of a project with the use of natural processes and the creation of a functioning ecosystem. Adaptive project development and management are important elements, with room for experimentation to further optimise techniques. With traditional structures there is a tendency to use commonly applied techniques to reduce risk and minimise construction time and costs, such as through re-using materials or cost-effective timing that may be combined with other projects.

Build and implement

The construction materials and timing are important for the success of nature-based methods. Natural materials that are site-specific should be used as far as practicable. For example, the grain size of sand for a beach renourishment should be relevant to the aeolian sand transport potential to maximise dune building. If there is not a natural seed stock, the materials needed can include sourcing of plants, or shellfish from aquaculture. The construction may need to be timed with reproductive events to maximise the chance of recruitment following the placement of settlement substrate. Similarly, timing may also be important in minimising the ecological impact of construction. Continued consultation with experts in the functioning of the ecosystem of interest will ensure the choice and placement of material is appropriate for achieving the ecological objectives. Flexibility during the construction process will allow for optimisation of methods and changes based on environmental conditions, or stakeholder needs (Case study: Shellfish reef breakwater).

Step 4: Operation and Maintenance

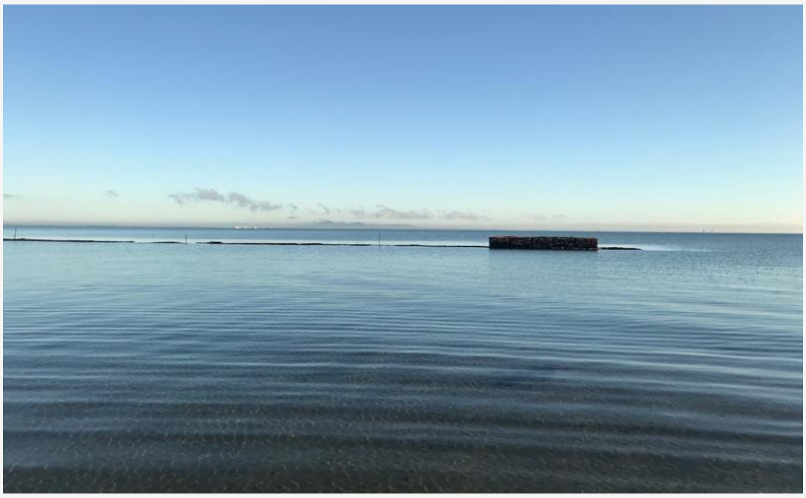
Monitoring is needed to ensure that the project retains its effectiveness and develops as expected through time. It also adds to the evidence base for nature-based methods and can be used to inform future projects. This will improve identification of key performance factors, which would reduce future monitoring costs and provide additional guidelines for designing nature-based methods aimed at risk reduction. The organisation responsible for monitoring and management should be defined early in the project. The purpose of monitoring of a traditional structure is to ascertain if its structural integrity and functioning is being maintained for the duration of its set design life. Nature-based methods, in contrast, are expected to develop incrementally and be adaptive (i.e., have an ongoing design life), as long as the ecosystem has the time, space and resources to respond to any environmental changes. Monitoring is thus important to determine when project objectives have been met and if they are being maintained through time.

Monitor, evaluate and adapt

Protocols for monitoring nature-based methods have been developed and should include measures that assess the ecological, engineering and socio-economic objectives[428, 429] (Appendix 1). Monitoring and evaluation programmes can be qualitative, semi-quantitative or quantitative, depending on the skills and time available. For example, visual techniques such as taking photographs take low effort and limited time and can be used to initially document project success or problems to address. Survey data on the structural components of the hard or natural elements of the project take more time to collect (at least a year) but will inform if a project is initially meeting its goals. Evaluation based on a structured experimental design of the target function would require several years, however, it provides high precision data that will be useful to inform future projects. This may be particularly important when using methods that have yet to be widely applied. The monitoring should highlight what did and didn’t work and why, to inform follow-up actions to adaptively manage a project. This may include maintenance or additional interventions. Results should be shared publicly to enable improvement in best practice for the application of nature-based methods.

Box 3.1 Case study: Shellfish reef breakwater, City of Greater Geelong, Victoria

A shellfish reef breakwater was constructed in 2018 at Ramblers Road, Portarlington in Port Phillip Bay to manage foreshore erosion and flooding of adjacent properties. The breakwater was constructed of modular steel cages filled with recycled basalt for one half of the breakwater and a mixture of recycled shell (mussel, oyster, scallop) and rock in the other half of the breakwater. The breakwater was then seeded with mussels from a nearby farm to accelerate the establishment of a shellfish reef. The breakwater was initially designed to be emergent with two tiers of units, but during construction of the top tier residents became concerned about the aesthetics of the breakwater, as it could be seen during all tides. The top tier of the breakwater was subsequently removed and placed onshore of the bottom tier, creating a wider but lower-crested structure. The wave attenuation of the structure was not compromised as the increase in width compensated for the loss in structure height. This adaptive redesign resulted in greater stakeholder support and higher ecological value through increased space available for mussel colonisation. This project is also exemplary in the incorporation of experimentation into the design, with the testing of different materials and mussel seeding methods on shellfish reef development.



Shellfish reef breakwater with two stacked modules during construction

© Ralph Roob

Shellfish reef breakwater with two adjacent modules after construction

© Ralph Roob

4 Benefit-cost analysis of nature-based methods

Benefit-cost analysis (BCA) is an economic decision support tool that enables decision makers to understand the trade-offs between investing in different project, program or policy options. BCA aims to identify the social worth of the options that are being compared with one another[430]. This is achieved through quantifying the benefits or gains in utility or wellbeing, and the costs or losses in utility/wellbeing. Gains and losses are monetised; that is, they are measured in terms of an individual's willingness to pay to receive more of a good or service, or willingness to accept compensation to go without it.

A BCA produces decision metrics that provide guidance about the net impact that a project could have on the wellbeing of the relevant population. These metrics enable decision makers to prioritise projects for investment in a way that maximises the overall benefits, given the set of projects being considered[431].

In the context of nature-based methods, the value of such an application is obvious. Coastal managers have limited financial resources to invest, but have multiple impacts to manage over multiple locations, and multiple solutions available for management. BCA offers the ability to transparently distil which options, including comparisons across nature-based and hard infrastructure solutions, will make best use of those resources.

Many texts and Government guidelines exist providing guidance on how to conduct BCA in general and it is not the intent here to repeat this advice (see Box 4.1 at the end of this section for a list of examples). This Section instead provides a brief summary of the key steps and their relevance to decision making about nature-based coastal defences.

Step 1: Defining the Project

The initial phase of a BCA is to clearly define the set of alternative projects to be compared. The information required for this should be gathered as part of the 'Initiation' and 'Planning and design' phases (see Section 3: Implementation Framework). In preparing this information for the BCA, it is important to think about a number of factors.

Project scope and the baseline

First and most importantly, the baseline scenario (i.e., the 'without project' scenario) must be defined. In a BCA, the objective is to compare the benefits and costs of a project – the 'with project' scenario – to the benefits and costs of continuing in a business-as-usual case, or the 'without project' scenario.

The scope of the BCA must be defined for each of the projects. This refers to which benefits and costs are relevant to include in the assessment. In the case of nature-based methods, the full range of 'co-benefits' (see Section 1.3: Benefits of Nature-Based Methods) should be considered as part of this scope, and this will mean that the range of benefits and costs will extend from market-based to non-market based, discussed further in Step 3 below.

Whose preferences matter?

While identifying the relevant benefits and costs, it is good practice to identify from the outset who the stakeholder groups are, such that the proportion of benefits and costs borne by each group can be attributed. Establishing who gains and who loses from a project will help with subsequent discussions regarding how equitable the distribution of benefits and costs are. Also, identification of the beneficiaries can potentially reveal who should pay for a project. Defining whose benefits and costs matter is relevant for setting the geographic scale, for example, whether the BCA is conducted at a scale relevant to a regional, state or national community.

Timeframes

The timeframe for comparing benefits and costs is important for a number of reasons. As outlined (see Section 1.4: Considerations for their Use), nature-based methods are best suited to situations where forward planning is possible; that is, in situations where buffer zones still exist, and there is time to implement a nature-based solution and allow it to mature before hazard risks become too great. This will generally mean that, while the costs of implementation may be immediate, the timeframe for the benefits of a project to be realised will be longer (also see the discounting step below).

The 'with project' and 'without project' scenarios must be compared over an equivalent timeframe, and all project scenarios that are being compared should be compared over the same timeframe. This means that the project with the longest timeframe required to deliver all costs and benefits being considered should be the reference point over which all projects are assessed. Note, however, that it is generally not worthwhile assessing project costs and benefits beyond about 50 years, as the principle of discounting means these values become very small, and there is often a high degree of uncertainty about the magnitude of benefits and costs beyond these timeframes.

The timeframe is critical for defining the 'without project' scenario. The key consideration here is that the 'without project' scenario, while reflecting a business-as-usual situation, may not be the same thing as current conditions, as those conditions are likely to change over time. The importance of this is easily illustrated in the context of nature-based defence: the current condition for a beach may be that there is minimal beach erosion that does not negatively affect infrastructure or significantly reduce recreational access. The future condition in 'Year X' may be that without any intervening project, roads, houses and other facilities may be lost and the beach becomes too dangerous to access. If the timeframe for the BCA is to make an assessment of benefits and costs through to 'Year X', then the impacts occurring under the latter would define the appropriate baseline, in terms of what would happen 'without' intervention, and this obviously implies a very different magnitude of costs compared to current conditions.

Step 2: Identifying Biophysical Impacts

For each positive and negative impact that could result from a project, an assessment needs to be made about what the magnitude of the impact will be. For nature-based methods, this will require there to be an established understanding of the biophysical processes and how different interventions will alter these processes. In turn, this needs to be linked to the impact a project can have on the assets or values being considered in the BCA, for example: how many houses, kilometres of road, or kilometres of beach might be protected; how many hectares of fish habitat are restored; or, how many tonnes of carbon are sequestered?

The timeframe for specific impacts must be captured here too. This could be different (i.e., less than) the overall timeframe for the analysis: some impacts may occur almost immediately, while others may take time to establish. It is important to know for Step 4 when a benefit or cost will be realised.

Step 3: Valuing Impacts

Once the biophysical impacts are ascribed to a project, they need to be monetised. The valuation process is based on the concept of willingness to pay, where an individual's gain in wellbeing or utility is measured by how much of their wealth they are willing to trade to secure a unit of a good or service[432, 433]. In theory, wealth (and what is traded) could reflect any tangible items that belong to the individual, but this is generally taken to mean an individual's tradeable resources (typically, disposable income). For market-based benefits and costs, willingness to pay will typically be reflected by the market price for a unit of a good or service. For non-market benefits, alternative approaches are required to estimate willingness to pay. Non-market benefits are an important consideration of nature-based methods and are discussed in some detail below.

What is a benefit and what is a cost?

An important consideration in valuing impacts is whether certain values should be considered as benefits or costs. All positive outcomes of a project, relative to the baseline scenario, are considered benefits. In the context of coastal defence, benefits include the damage costs avoided by protecting infrastructure.

Costs can reflect a number of things, including:

1. the capital project costs, which are typically drawn from the (limited) pool of investment cash that will fund the selected project(s);
2. project operating costs, such as maintenance or monitoring costs, which might also be drawn from the investment pool (at least for the early years of the project);
3. in-kind costs provided by the managing organisation, such as in-kind labour, which are drawn from a separate pool of funds to the cash investment pool, but are often similarly limited as they are capped by the organisation's capacity to contribute resources;

4. negative externalities, which are the negative spin-offs that a project might generate; and
5. the opportunity costs of investing in one project over investing the money in something else (dealt with by discounting, see Step 4).

In the case of nature-based defence decisions, there will likely be many negative externalities to consider dependent on the projects being compared. A good example is if a beach nourishment project is being compared to construction of a seawall to manage erosion. A range of costs of the seawall could include materials, labour, and loss of access to recreational space. The materials and labour would be paid out of the pool of funds and represent the capital project cost. The loss of access to recreational space is a negative externality. The origin of benefits and costs becomes important in the calculation of the benefit-cost ratio (see Step 5 for a discussion of why)[434].

Project costs

The project costs – both capital and operational expenses – are typically the simplest to estimate. These are market-based costs that will reside in the denominator of Equation 4.3 below. They should include all of the costs that will be drawn from the investment funds to implement the project, for example the cost of labour, materials, construction and any monitoring and maintenance costs that might be relevant while the nature-based defence project is being established. Generally, the project proponent will have this information on hand.

Market benefits and avoided costs

For nature-based methods the foremost benefit to include is that of the assets protected from coastal hazards. This could be quantified, for example, as the avoided replacement costs of infrastructure and housing.

Any market-based co-benefits (defined in Section 1.3: Benefits of Nature-Based Methods) should also be included. Benefits (and potentially negative spin-offs) that are market-based are evaluated based on how much the project changes consumer surplus and producer surplus. Estimation of supply and demand curves enables calculation of this surplus, and guidance on how to do this is provided in standard micro-economic theory texts[435].

In practice, for co-benefits of nature-based projects this will often focus on estimation of producer surplus (i.e., profit), because we would anticipate individual projects to contribute small quantities of goods (e.g., a small additional fish biomass) relative to the overall market (the total existing biomass in the commercial fishery). That will mean the price per unit of the good is unlikely to change, and consumer surplus will remain constant, for example: an individual consumer purchases and enjoys a fish in the same way as before, such that the only change in surplus is additional profit made by the producer for the additional quantity sold.

Non-market benefits

It is reasonable to assume that nearly all nature-based defence decisions will have some form of public impact, whether that be through the impacts on public assets, including availability of recreational spaces and facilities, or through the changes that could improve natural spaces and ecosystems. This in turn will lead to changes in welfare or utility of local, and sometimes broader, communities. Prioritisation of projects in a BCA should therefore encompass community preferences for non-market values, particularly those environmental and social co-benefits (defined in Section 1.3: Benefits of Nature-Based Methods).

In economic valuation, the intangible social and environmental outcomes of a project are referred to as 'non-market values'[431]. They include use-related values such as the value drawn from recreation, use of amenities, or pleasurable aesthetics of a coastal location. They also include non-use values, which encompass the existence value an individual gains through knowing, for example, that a rare species or ecosystem exists and is being protected.

Non-market values are not traded in a marketplace, and so they do not have a monetary value in the same sense that other marketed goods and services do. A BCA, however, requires that all benefits and costs are measured in a consistent metric, and in practice this is a monetary metric.

This means we need to estimate a monetary-equivalent value for the non-market benefits to be included in a BCA. The theory exists to enable this, and it is consistent with the theory underpinning how we estimate market-based values, in terms of understanding people's willingness to pay for a benefit.

A suite of non-market valuation methodologies exist to estimate how much people are willing to pay for improvements in social and environmental outcomes. These are well documented and widely applied in environmental valuation[436]. Depending on the type of benefit to be measured, there are two primary methods that can be used: revealed-preference; and stated-preference methods.

Revealed-preference methods are based on observations of people's behaviour. Recreational benefits, for example, can be evaluated by observing how far people travel to visit a location for recreation, and using the distance from their home to infer their travel costs (the travel cost method[437]). This provides an estimate of the minimum amount people are willing to pay for a recreational benefit. For nature-based methods, this approach would be relevant for measuring such benefits as: maintaining beach access; providing sheltered lagoons to swim in; providing new reefs to surf, snorkel, dive or fish on; and so on.

Hedonic pricing methods are another revealed-preference approach[438]. Hedonic pricing utilises existing markets (e.g., housing markets) to reveal how the non-market characteristics of a good (a house) influence its market value. For example, houses close to the coast may have a positive amenity value for their proximity to coastal facilities and seascapes, which generally attracts a premium to the house price. However, coastal hazards present a negative externality, where some houses can lose value because of the increased susceptibility to erosion, storms and flooding[439].

Revealed-preferences can only estimate the values associated with people's use of public assets. The environmental benefits of nature-based methods will often mean that non-use values are important too; for example, the values people place on the existence of marine wildlife. These values are not always associated with an interaction between an individual and the thing that is valued, so we cannot observe behaviours. Instead, we can use stated-preference techniques – a set of survey-based methods – where we can ask an individual about their willingness to pay to achieve an outcome.

Two common stated-preference applications are contingent valuation and discrete choice experiments[440]. Contingent valuation approaches focus on understanding how much people would be willing to pay for an overall program or policy change[431]; for example, how much an individual would (hypothetically) be willing to pay for a nature-based defence that would prevent erosion of a local beach.

Discrete choice experiments break a policy or program down into the different features or attributes that define it, and present different combinations of these attributes as a set of hypothetical options. The options contain different levels of each attribute, such that the quantity or quality of the attributes varies across the options. One of the attributes is the hypothetical cost of the option to the survey respondent. The respondent has to trade-off the changes in the attributes, including the cost, across the set of options, and vote for their preferred option. Through statistical analysis of respondents' choices, the analyst can estimate how much people are willing to pay for marginal improvements in each of the attributes[441]. This approach can be used to distil what specific benefits of a nature-based method are most valuable. For example, the attributes can be defined in terms of the various non-market outcomes a project could generate such as kilometres of sandy beach available for recreation, protection of public recreational amenities, improved safety for people using the beach, hectares of foraging habitat created for marine fauna species, and so on.

A set of publicly available tools have recently been developed for coastal managers to implement non-market valuation surveys and collect data on community values for assets affected by coastal hazards[442]. Travel cost and discrete choice experiment approaches are embedded within the survey templates, providing a starting point for estimating some of the non-market benefits associated with nature-based defences.

However, it is not likely to be practical to conduct a non-market valuation study to estimate all of the non-market benefits that should be considered in a BCA for a nature-based method, every time that a BCA needs to be conducted. Benefit transfer, a process that extrapolates non-market values from existing studies to predict values for a new project, is a useful tool when the budget and timeframe cannot support conducting an original valuation study[443, 444].

There will be a trade-off between the accuracy of the data collected via an original study and the efficiency of using benefit transfer instead. Rogers et al. (2019)[439] provide a commentary on the state and availability of the non-market valuation literature for natural hazards (more broadly than just coastal hazards), with the perspective of how readily these values can be used in benefit transfer. They conclude that original studies should be conducted when feasible but using less accurate data from benefit transfer is still likely to be a better alternative than omitting non-market values from a BCA altogether. The latter makes an implicit assumption that these values are zero, and it follows that they then do not vary in magnitude between decision options. For nature-based defences, this is almost certainly an incorrect assumption.

Avoid double-counting

There should be no double-counting of the benefits or costs included in the analysis. Double-counting errors often occur when there is a mixing of benefit categories, such as the inclusion of a causal process (hectares of fish habitat restored) and the end-state value (the increased utility that a recreational fisher receives from fishing at the restored site)[445]. Practitioners could use an estimated value for either category in the BCA (e.g., an individual's willingness to pay per hectare of habitat restored *or* willingness to pay per fishing trip), and this decision will be guided by the availability and accuracy of data for each category, but they should not include values for both in the analysis. Similarly, the context in which causally-linked categories are evaluated needs to be considered with the risk of double-counting of co-benefits in mind. For example, an estimate of willingness to pay per hectare of habitat might encompass an individual's value of that habitat to provide recreational fishing benefits, to improve water quality for swimming, *and* to provide habitat for a threatened species.

Capture project risk

Project risk in the context of BCA refers to the probability that a project will not deliver, in full or part, the anticipated net benefit. This is different to uncertainty, which is about incomplete information, and dealt with via sensitivity analysis. Multiple risks should be considered in a BCA, and the values of benefits and costs should be adjusted accordingly.

Systematic risks are risks that arise from unforeseen macroeconomic conditions, beyond the ability of the project manager to control[433]. These risks can result in the positive value of project outputs being smaller than anticipated. Systematic risks can be accounted for by adding a premium to the discount rate, or by scaling down the projected benefits of the project in recognition that they might be over-estimated. Nature-based methods, and environmental projects generally, are likely to carry high systematic risks given their relationship with global challenges such as climate change.

Other risks are specific to the project[434]. For nature-based methods they can include things like risks of technical or implementation failure, for example, where the biophysical design elements of the project do not go to plan, or where the governance arrangements lead to poor management of the project. There can also be financial risks of procuring sufficient budget. While the upfront investment budget for capital and shorter-term operating expenses may be certain, some nature-based coastal defence projects may require on-going or ad hoc monitoring and maintenance budgets, for example to repair hybrid structures following a storm event. Even when factored in as part of the pool of investment funds, the availability of longer-term operating budgets is often not guaranteed.

Values can be risk-adjusted using a probability distribution and a Monte Carlo simulated analysis[446]. Where available data or expert judgement can reasonably identify the probability that a benefit is likely to be achieved, a simpler approach can be to specify the risk of an outcome failing, *R*. The expected value of the benefit can then be multiplied by $1 - R$ [447], based on the simplifying assumption that there is a binary distribution of project outcomes: success and failure, with failure providing zero benefits. For example, assume that a dune restoration project is expected to avoid damages of $1 million to built infrastructure. There is an 80% probability that the project will succeed in doing this. The risk-adjusted benefit for protecting the infrastructure is then *$1 million x (1 – 0.2) = $800,000.*

Step 4: Discounting

Future benefits and costs must be discounted to calculate a present value. The practice of discounting reflects the opportunity costs of investing in a given project (relative to using funds to invest in a different project), and that people prefer to enjoy benefits now rather than waiting to enjoy them in the future[435]. This practice is clearly important for nature-based defences, given the majority of costs are borne upfront in the implementation phase, and the benefits may take years to fully realise.

The appropriate discount rate to use has been debated widely in the economics literature, including whether a social or market discount rate should be used, whether the rate should be constant and how big the rate should be[433, 448, 449]. As nature-based methods will typically generate public benefits, a social discount rate is likely to be the most appropriate. In Australia, the Office of Best Practice Regulation, Infrastructure Australia and nearly all State Governments suggest a default social discount rate of 7%, using between 3% and 10% in sensitivity analyses (see Pannell 2019[434], p26 for a summary of recommended rates by Australian jurisdiction). Standard practice varies internationally, for example, the UK Treasury Greenbook recommends a rate of 3.5%. The current low interest rates (globally) would support using a lower discount rate, though we cannot be certain how long low interest rates will remain. This points to there being uncertainty about what future discount rates will be, and there are arguments for using a discount rate that reduces over time which allows for this uncertainty[449]. One thing clear from this debate is that it is critical to test the discount rate in the sensitivity analysis to understand how adopting different rates might affect the decision outcome.

On the assumption that a constant discount rate is being used, the present value of all benefits and costs is calculated according to Equation 4.1, where *X* is the value of the future benefit (cost), *r* is the discount rate at time *t*, and benefits that occur in each year *t* are summed over the project life, *T*:

$$Present\ value = \sum_{t=0}^{T} \frac{X_t}{(1+r)^t} \qquad \text{(Equation 4.1)}$$

Step 5: Decision Metrics

Two decision metrics are produced in the BCA. The first is a measure of net present value (NPV), which identifies which projects are worthwhile on the basis that the benefits are greater than the costs. The second is the benefit-cost ratio (BCR), which identifies how much of a benefit you gain per dollar invested.

NPV is the sum of the present value of benefits less the sum of the present value of costs (Equation 4.2):

$$NPV = \sum present\ benefits - \sum present\ costs \qquad \text{(Equation 4.2)}$$

The BCR is the sum of the present value of benefits divided by the sum of the present value of costs (Equation 4.3):

$$BCR = \frac{\sum present\, benefits}{\sum present\, costs} \qquad \text{(Equation 4.3)}$$

We note that some BCA texts and guidelines advocate for using only the NPV to rank and prioritise projects for selection[432, 435]. This is because of a misconception that the BCR can give inconsistent information depending on where one places certain costs*, and that this could change the advice given when comparing projects. However, costs in the denominator should reflect only the costs drawn from the pool of resources being administered by the project organisation to fund the project. As Pannell (2019)[434] explains, other costs – such as costs that other businesses or individuals might bear – do not constrain project selection, and therefore should reside in the numerator†. Of the costs identified in Step 3 above, the capital project costs (i) belong in the denominator, as do the project operating costs (ii) if they are part of the investment budget. Any in-kind costs (iii) that are born by the project organisation can also be placed in the denominator‡. It is unambiguously clear that any negative externalities (iv) belong in the numerator.

Provided this rule is applied, the BCR will provide consistent advice when being used to compare projects. The two decision metrics should then be used as follows to guide ranking and prioritisation of projects:

- Use NPV or BCR: if there is an unlimited pool of resources for implementing projects (an unlikely scenario). All projects with a NPV>0 or a BCR>1 should be implemented, as these projects generate a net benefit.
- Use NPV: when you are comparing a set of mutually exclusive projects meaning that selecting one project rules out selecting another. For example, comparing a project to construct a shellfish reef using a natural shell substrate with a project to construct a shellfish reef (in the same location) with limestone substrate would be mutually exclusive. In this case, select the project that has the largest NPV. If there

* For example, assume a seawall is proposed where the cost of construction is $50,000 (clearly a cost) and highway infrastructure is protected worth $200,000 (clearly a benefit). The seawall will reduce public access to the beach, resulting in lost recreational value of $100,000. If this $100,000 is included (incorrectly) as a cost in the denominator, the BCR = $200,000 / $150,000 = 1.3. If it is included (correctly) as a negative benefit in the numerator, the BCR = $100,000 / $50,000 = 2.

† Dobes et al. (2016)[360] suggest using a 'net benefit investment ratio', rather than a BCR. In this ratio, they specify that only the capital or investment costs be included in the denominator, while operating costs be included in the numerator. Often, operating costs (at least in the short term) of a project are factored into the investment budget and should be included in the denominator of the BCR, but in cases where only capital costs are funded by the investment pool then the net benefit investment ratio is the same as the BCR..

‡ Theoretically, the in-kind costs come from a separate pool of resources to the cash investment pool, and an optimisation model is required to rank projects with complete accuracy when there are multiple budget constraints involved. However, the resources (cash and in-kind) are all being constrained by the project organisation. Inclusion of these costs in the denominator has been found to provide a good approximation of the ranking outcomes that would be derived by a theoretically correct mathematical optimisation model[348].

is a budget constraint, first establish which projects you can afford given your budget, and then among those select the one with the biggest NPV.

- Use BCR: when there is a budget that is constrained but that can be exhausted and you are comparing a set of distinct projects, where it is possible to select more than one project (i.e., they are not mutually exclusive). For example, comparing a project to restore a seagrass meadow in Perth with a coral reef in Cairns would be distinct. In this situation, you can use the BCR to rank projects and then select as many of the best-ranked projects as you can within the available budget. In this situation, NPV does not rank the projects correctly.

Readers are referred to Pannell (2019)[434] and the online Pannell Discussions[§] resource for a three-part series explaining appropriate use of these decision metrics and correct placement of costs in the BCR.

Step 6: Sensitivity Analysis

Often there is uncertainty associated with some of the data inputs in a BCA. This can range from what the appropriate discount rate is, through to how much confidence there is in the estimated dollar values of benefits and costs (especially for non-market values), and our certainty in the biophysical impacts that different projects will generate. There is also the risk of 'optimism bias', where project managers and analysts tend to over-estimate the benefits and under-estimate costs[446].

Sensitivity analyses address these uncertainties and provide guidance as to how robust the decision outcome of the BCA is to changes in the values of certain variables. It provides an additional layer of information for the decision maker: imagine a scenario where Project A and Project B are closely ranked in terms of their NPV, with Project A providing a net benefit of $2 million, and Project B a net benefit of $1.8 million. The budget available can only support one of the projects, and this outcome would suggest that the budget be invested in Project A. After a sensitivity analysis it is revealed that the NPV of Project A is highly variable dependent on the assumptions made about the value of benefits included, with NPV potentially ranging from $1 million to $2.3 million. The benefits of Project B are more readily predicted, and NPV ranges from $1.7-$1.9 million. In this situation, the decision maker can weigh the risk of Project A not delivering on its anticipated $2 million of benefits and consider whether Project B is a better prospect given the certainty of realising its (still substantial) proposed benefits.

Pannell (1997)[450] outlines the theory and practice of sensitivity analysis for economic decision making, including discussing its various uses from the more obvious input to guide decision making through to how it can help improve communication, understanding of system relationships and model development.

It is common practice, as a bare minimum, to test sensitivity to the discount rate. As discussed above, most Australian Treasury departments will recommend testing the

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[§] https://www.pannelldiscussions.net/2019/08/322-npv-vs-bcr-1/

robustness of the NPV or BCR to a range of discount rates, roughly set at 3%, 7% and 10%.

For benefit and cost variables, values can be changed to reflect the confidence in the estimate used in the analysis. This can be done for one specific variable at a time, or simultaneously for multiple variables using a Monte Carlo analysis to estimate a probability distribution for the NPV[432].

Our scientific understanding of nature-based methods is still emerging, meaning that sometimes we may have such a poor understanding of the real value of a particular benefit that our confidence interval is too wide for it be usefully identified. In this case, we can use the sensitivity analysis to identify where the 'break-even' point is. That is, by how much does the value of the benefit need to change before the decision outcome changes from a favourable (NPV>0, BCR>1) to an unfavourable one (NPV<0, BCR<1). This information enables the decision maker to make a judgement about whether the break-even point sits within what is a plausible range of values for that variable, and subsequently gauge the importance of the variable. It can identify variables that are particularly sensitive, where a small change in the value changes the decision outcome. This information guides future research efforts in determining for which variables it is a priority to gather better data[446].

Further Considerations

The advice provided above should be supported through consultation of other standard guidelines for conducting BCA in Australia, where further information on the general requirements and caveats of BCA to satisfy economic theory are discussed in much more detail (refer to Box 4.1). The advice above draws attention to particular aspects of the BCA process where additional care should be taken for nature-based methods. Following the discussion above, it is worth reflecting on the various value-added benefits that a BCA can provide for guiding nature-based defence investment decisions, beyond simply perusing the NPV and BCR decision metrics.

In particular, the first steps of conducting a BCA, where one defines the scope of proposed projects, the without-project baseline, the range of anticipated biophysical impacts that the project(s) will have and the timeframes for those impacts is, of itself, a useful process to undertake. The information required to complete this process aligns to the implementation framework (see Section 3). However, it must be prepared in a very structured way for integration in BCA, which often leads to seeking a level of clarity and transparency – as to what the decision is and its implications are – that is otherwise not achieved.

The sensitivity analysis stage of a BCA can also be illuminating. This process can identify whether benefits for which we have a high degree of uncertainty (in terms of their expected value) have a sensitive relationship with the decision outcome. In essence, the BCA can not only be used as a tool to prioritise investments in the projects being considered, but also as a tool to prioritise research needs for gathering more data.

Within the BCA process, the distribution of benefits amongst stakeholders can be assessed. This makes it possible to identify who the winners and losers of a decision will be. From an equity perspective, this information is useful, particular to identify whether marginalised or minority groups will be adversely impacted. With this in mind, we stress that BCA as a process does not claim to identify the best projects to fund based on equity implications: it does not distinguish between the value of a dollar to individuals with low or high incomes[432]. It is purely a decision support tool and subsequent steps in the decision process need to take place following the economic prioritisation to consider implications such as social equity.

As outlined above, quantification and inclusion of non-market values for prioritising investment in nature-based methods will be key, particularly when the investment is supported by public funding and the breadth of community benefits should be considered. Indeed, the costs of undertaking a nature-based project in some instances may exceed the market-benefits, such that without including the non-market benefits in the BCA a project with significant community benefit might not be prioritised correctly. In the case of private sector investment, quantification of these values may still be important. While the financial bottom line is likely to drive decision-making in this case, among projects that are financially viable (market-benefits exceeding costs), inclusion of non-market values can help investors to determine which projects provide added benefits that will improve their corporate standing and social licence to operate.

Historical debates have questioned the relevance of including data estimated via non-market valuation in BCA[436]. However, best practice in conducting non-market valuation has advanced significantly in recent decades[451], improving the accuracy of the techniques, and the dangers of omitting this information from a BCA could be significant[439, 447]. Globally, Bateman and Kling (2020)[452] review the important historical role that non-market valuation and BCA have contributed to key environmental decisions in the US, UK and the EU. Nationally, the Productivity Commission provides guidance on the use of non-market valuation in support of its inclusion in BCA[453]. Further, Rogers *et al*. (2015)[436] show that environmental decision makers in Australia are receptive to the inclusion of non-market values in the use of decision support tools, including in BCA, for evidence-based decision making.

Finally, it is worth noting that other tools do exist to provide decision support for prioritising investment. The most common alternatives to BCA are multi-criteria analysis and cost-effectiveness analysis. Multi-criteria analyses identify expected impacts of a project, and then assign subjectively determined weights to quantify their importance. Dobes and Bennett (2009)[454] critique this approach, acknowledging that the subjectivity involved ultimately limits the robustness, replicability and transparency of how decision outcomes are arrived at. Cost-effectiveness analysis avoids the need to monetise benefits by assessing the cost of a project per unit of ‘effectiveness’[432]. For example, this could be expressed as the cost per kilometre of beach protected. This approach will have limited application for prioritising nature-based coastal defence investments as it focusses on identifying effectiveness with respect to a particular outcome. That is, it will not be inclusive of the full range of co-benefits, which will commonly be a core part of justifying the net benefit of, and subsequently investment decisions in, nature-based methods (see Section 5: Enablers). Relative to these other prioritisation tools, BCA should provide a more comprehensive and objective approach to prioritising investment in nature-based methods.

Box 4.1 Examples of existing guidelines for benefit-cost analysis

Boardman, A., Greenberg, D., Vining, A. and Weimer, D. 2014. Cost-Benefit Analysis Concepts and Practice. 4th Edition, Cambridge University Press, Cambridge.

Commonwealth of Australia 2006. Handbook of Cost Benefit Analysis January 2006. Department of Finance and Administration, Canberra.

Dobes, L., Leung, J. and Argyrous, G. 2016. Social Cost-Benefit Analysis in Australia and New Zealand: the state of current practice and what needs to be done. Australian National University Press, Canberra.

Hanley, N. and Barbier, E. 2009. Pricing Nature: Cost-Benefit Analysis and Environmental Policy. Edward Elgar, Cheltenham.

NSW DPIE 2018. Guidelines for using cost-benefit analysis to assess coastal management options. Department of Planning, Industry and Environment, Parramatta, NSW.

OECD 2018. Cost-Benefit Analysis and the Environment: Further Developments and Policy Use. OECD Publishing, Paris.

Pannell, D.J. 2019. INFFEWS Benefit: Cost Analysis Tool: Guidelines. Cooperative Research Centre for Water Sensitive Cities, Melbourne.

Pannell, D.J. 1997. Sensitivity analysis of normative economic models: theoretical framework and practical strategies. Agricultural Economics, 16: 139-152.

Wise, R.M. and Capon, T.R. 2016. Assessing costs and benefits of adaptation. CoastAdapt Information Manual 4, National Climate Change Adaptation Research Facility, Gold Coast.

5 Policies and legislative settings relevant to nature-based methods

The potential for increased acceptance and uptake of nature-based methods and the applicability of these guidelines sits within a broader context of coastal management and adaptation law and policy in Australia. This section was informed by a total of 12 interviews with policy makers in high level management positions in strategical coastal planning for state and federal governments (in the following jurisdictions: Victoria; Queensland; Western Australia; Tasmania; South Australia; and the Commonwealth). This section describes the current policy landscape within which nature-based methods may be implemented, finishing with a summary of the potential barriers and enablers to their uptake.

Key Policies for Coastal Management

Federal government policy

Under Australia's Constitution, state governments have decision making power over the coastlines, their development and management[455]. Therefore, Australia's federal government, governed by the Constitution, has limited capacity to make decisions about how Australia's coastlines are managed with the exception of:

- Coastal environments that are classified as Ramsar wetlands
- Coastal environments that contain a threatened species
- World Heritage listed areas such as the Great Barrier Reef
- Coastal lands used for defence purposes
- Commonwealth run National Parks that contain a coastal environment[**]

Notwithstanding this, the federal Government does hold financial and funding powers over the state governments, and this can influence the decision-making processes of the State Government and shape coastal management priorities. The main focus described by the federal government representative in the interviews was the provision of information and research on current and future priorities for climate change adaptation (sea level rise, marine heat waves, risks for coastal environments etc.). An important role, therefore, of the federal government in nature-based methods is prioritising research in this area to support state and territory policy.

At a Federal level, Infrastructure Australia (a statutory body established under the Infrastructure Australia Act 2008) has listed a national coastal hazards adaptation strategy

[**] Commonwealth run National Parks are a very small minority of Australia's national parks and conservation reserves. National parks and reserves are run by state governments under the Australian Constitution.

as a high priority initiative on the Infrastructure Priority List[456], including nature-based methods as an infrastructure option for adaptation.

State and territory government policy

Policy for nature based coastal defence sits within broader governance frameworks for coastal management and climate change adaptation at a state/territory level. Although thematically similar, coastal governance varies from state to state. This means that for each state, a different framework of legislation, policies, strategies and plans will guide decisions about coastal protection and the potential implementation of nature-based coastal defences (Table 5.1).

Table 5.5. Key legislative and policy frameworks identified to support decisions in coastal management and risk reduction.

JURISDICTION	LEGISLATION	POLICY	STRATEGY/GUIDELINES
Federal	Environmental Protection and Biodiversity Act 1999		
New South Wales	Environmental Planning and Assessment Act 1979 Coastal Management Act 2016*	State Environmental Planning Policy (State and Regional Development) 2011 State Environmental Planning Policy (Coastal Management) 2018	Coastal Management Manual
Northern Territory	Aboriginal Land Rights (Northern Territory) Act 1976		Coastal and Marine Management Strategy 2019-2029
Queensland	Coastal Protection and Management Act 1995	State Planning Policy 2017 Queensland Coastal Plan**	
South Australia	Coast Protection Act 1972	Coast Protection Board Policy (revised 2016)**	
Tasmania	State Policies and Projects Act 1993	State Coastal Policy 1996	
Victoria	Marine and Coastal Act 2018	Marine and Coastal Policy 2020*	Marine and Coastal Strategy 2021* (*in prep.* due mid 2021)
Western Australia	Planning & Development Act 2005	State Coastal Planning Policy 2.6 – Coastal Planning	State Coastal Planning Policy Guidelines*** Coastal hazard risk management and adaptation planning guidelines***

Includes: *nature-based methods (as defined in Section1); **beach nourishment/dune management only; ***beach nourishment and artificial reefs only.

Policy provision for the protection of Australia's coasts generally follows one of two avenues. Coasts are protected either through policy explicitly developed to address the management challenges of coasts - for example through Victoria’s Marine and Coastal Policy. In other jurisdictions, coastal protection is provided for through planning policy, for example in New South Wales and Western Australia (Table 5.1).

Nature-based methods for risk reduction (as defined in Section 1 of this guideline) are explicitly referred to in two legislative or policy frameworks: NSW Coastal Management Act

2016 and VIC Marine and Coastal Policy 2020. In both of these documents, nature-based methods are inclusive of diverse marine habitats (e.g., coastal vegetation, biogenic reefs, dunes) and preference them as an adaptation strategy above traditional hard methods for protecting the coast. Other states – QLD, SA – preference what is called “soft solutions” over hard structures, this generally refers to beach nourishment and dune management methods only. In WA, beach nourishment and artificial reefs are listed as potential protection options, alongside the traditional structures. These options are not preferenced over traditional structures, and artificial reefs are considered as low crested breakwaters, with no reference to using them as a substrate to develop adaptive biogenic reefs (e.g., shellfish or corals). In the other jurisdictions – NT and TAS – nature-based methods are not included in the policies or strategies. In the Tasmanian policy it is stated that management of coastal hazards should minimise the need for engineering interventions, and in the Northern Territory marine strategy there is a dedication to climate resilience. Victoria is the only state that considers nature-based methods separately from protection options (refer to the hierarchy in Figure 1.1). This reflects the fact that nature-based methods are generally not designed to ‘hold the line’ in the traditional sense, but their value is that they are naturally adaptive to changes in climate, which will likely eventually include retreat to maintain coastal resilience.

Local Government Authorities and Committees of Management are responsible for the development and implementation of coastal management plans and land-use planning decisions, operating within the regulatory and policy frameworks established by the state or territory government, and therefore play a key role in the on-ground application of nature-based methods for coastal defence.

Barriers and enablers for nature-based methods

This section reflects the views of the interview participants, the majority of which were state government representatives whose main roles and responsibilities are, to design strategies to support policy development, work with local government to develop implementation strategies and interpreting policy to help guide planning outcomes.

Current policy

Despite the diversity in legislation and policies regarding coastal management across jurisdictions, the interviewees considered that the current policy landscape supported the implementation of nature-based methods, with particular reference to beach nourishment and dune management. It was viewed that there are already strong provisions and precedence for these options as there is a history of their use in Australia, and globally. However, the interviewees were less familiar with other nature-based options such as reef restoration and were less confident in the feasibility of those options within the current policy landscape. Some barriers to these options are outlined in the following paragraphs.

Barriers

Timeframes and risk

Participants considered that options such as reef restoration and coastal revegetation (e.g., saltmarsh, mangrove, seagrass) had less precedent than other options. When coasts are under significant and imminent threat, there is a lot of pressure to choose tested options with a shorter time to results. In fact, in areas that are at high risk already, nature-based methods are unlikely to be suitable for use in that area due to the need for time to establish (see Section 1.4 Considerations for their use). In areas that would be more suitable to nature-based methods, it was considered difficult to justify their application in an area that was not at immediate risk. This highlights the issues of current coastal management practices that are reactive to problems, rather than forward-thinking, which is in part exacerbated by how coastal protection is funded.

Funding

There are limited budgets for risk reduction on the coast, and this usually means that this funding is targeted towards existing high-risk problems with increasing pressure from changing risks. This raises the question of whose responsibility it is to pay for demonstration projects for less well-known approaches. Similarly, it should also be questioned as to whether funding spent now should all be focussed on high-risk areas or should we be managing for the future. For example, what are the cost (and environmental) savings of employing a nature-based method now versus a hard solution in a decade's time when the problem is exacerbated.

Jurisdictional issues

Management of the land-sea interface can often be complex and involve multiple organisations. The responsibilities for the terrestrial-based coastal options (e.g., beach and dune management) are often clearer and provided for within state strategic policy. However, management of marine systems (e.g., shellfish and coral reefs) for coastal hazard risk reduction is not as clearly accounted for amongst the many State agencies involved.

Political barriers

While nature-based methods may be provided for and prioritised as part of a preference for soft options in policies, they are not always chosen by local coastal managers because of political pressures from homeowners and developers. The decision about the on-ground intervention can be made by local or state government, or through a partnership between the two. Part of this political pressure may be due to stakeholders being less familiar with nature-based methods.

Precedent

This barrier is particularly relevant to managers closer to the implementation of potential nature-based methods (e.g., Local Government Authorities, Committees of Management etc). The barrier can be best described as the lack of simple and accessible operational precedents for the use of nature-based methods. When tasked with implementing infrastructure for risk reduction, many local jurisdictions will look to current practice and what can be accessed via experienced coastal engineering consultants and contractors. There is a lack of simple, standardised engineering practice (i.e., precedents) for nature-based methods done at scale showing coastal managers, decision makers and the broader community what can be done in Australia. This lack of examples of standardised, best practice for nature-based methods means organisations may default to current practice regardless of the relevant policy context.

Enablers

Co-benefits of nature-based methods

The primary enabler of nature-based methods identified by interviewees was the number of co-benefits that could be achieved with using these interventions over traditional methods (e.g., biodiversity benefit, water quality, carbon storage; see Figure 1.3). When effectively communicated to coastal managers, an evaluation of the added benefits can provide the business case needed to support their application. This highlights the importance of developing cost-benefit analyses for the different nature-based methods (see Section 4. Cost-benefit analysis for nature-based methods).

Summary

Key policy makers believe that the implementation of nature-based methods is accounted for within Australia’s current policy landscape, and thus policy change is not a priority. A major barrier is the funding and delivery of demonstration projects of those methods that have been less well-used to date, to provide precedent for their use. Providing this precedent can then expand the implementation of more diverse techniques for coastal risk reduction. This needs to be supported by an evaluation of the full suite of benefits provided by nature-based defences, which will further increase their cost-benefit. However, nature-based methods will still not be enabled if management of coastal risk continues to focus on ‘fixing’ high-risk areas rather than employing a holistic coastal adaptation strategy to support future resilience.

Next Steps

This document has provided the foundational guide to inform the national use of nature-based methods for coastal hazard risk reduction. The process of producing these guidelines has resulted in the identification of several focus areas to further enable the wider use of nature-based methods as an adaptation strategy in Australia:

- Design guidelines analogous to those that are well-established for traditional coastal protection structures need to be developed for different nature-based methods. Practical tools will need to be based on detailed studies of how the design of nature-based methods influences the surrounding hydrodynamic processes and coastal hazards. Design guidelines should also address how to maximise co-benefits of projects.
- Best-practice guidelines for working with traditional owners in managing coastal hazards and the integration of indigenous knowledge into the design and implementation of nature-based methods needs to be developed.
- A national spatially explicit mapping tool should be developed to guide what options/methods can be applied in a local setting. This is analogous to living shoreline decision support tools that have been developed for areas of the United States[457]. This could be integrated into a national coastal hazards adaptation strategy, as identified as a priority by Infrastructure Australia.
- In each state, a selection of demonstration projects should be supported to provide an exemplar and precedent for the community, and other decision makers. These demonstration projects if properly monitored can also contribute data to inform the development of design tools. An inventory of projects already using nature-based methods will also support design tools.

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Appendix 1

Table 1. Examples of criteria for measuring the success of nature-based methods (adapted from Morris *et al.* 2019[428]).

CATEGORY	GOALS: FUNCTIONS/SERVICES	MEASURE
Ecology	Native species biodiversity	At species level: species richness, biomass, abundance, % cover, percentage (%), community assemblage At habitat level: Habitat diversity At genetic level: Genetic diversity
	Invasive species	Number and abundance of species Ratio of native to invasive species
	Target species	Enhancement / recovery of abundance or survival of target species
	Ecological functioning and processes	Integrity of biological assemblage (e.g., functional groups) Biofiltration Water quality Primary productivity Ecosystem engineers / habitat-forming species Bioprotection Carbon sequestration
	Fisheries production	Enhancement of fisheries supply Usage of habitat / refuge by larvae
	Connectivity	Enhancement and/or reduction of connectivity
Engineering	Energy attenuation	Wave height reduction Current reduction
	Shoreline stabilisation	Horizontal shoreline location Sediment volumes
	Achieving adequate/ desirable sedimentation dynamics	Regulating sediment accumulation
	Reduce flooding	Surge extent/height
	Structural integrity	Structural integrity Durability/longevity Resistance to extreme weather events
Social	Aesthetics	Appeal to people
	Tourism and recreation	Waterfront accessibility People's awareness and use of the waterfront
	Education	People's knowledge and awareness of coastal biodiversity and eco-engineered shorelines

Continued on next page.

Governance and Policy	Facilitate use of eco-engineered shorelines	Funding incentives, permits, recommendations, regulations
	Hazard mitigation	Protection of property and life
Economic	Cost-benefit	Costs (e.g., capital project, operating, in kind) Market based benefits (e.g., avoided damage to buildings, and co-benefits such as carbon storage and fish production) Non-market based benefits (e.g., willingness to pay)

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