

Anticipated Climate Change Impact on the Coastal Protection Role Provided by Coastal Ecosystems in the Pacific and East Timor

Prepared for Department of Climate Change and Energy Efficiency Canberra, ACT

June 2011



Australian Government

Department of Climate Change and Energy Efficiency

AusAid



ISBN 978-1-922003-97-3

This report was supported by funding from the Australian Government under the Pacific Australia Climate Change Science and Adaptation Planning (PACCSAP) program.

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NIWA Client Report No:	HAM2011-017
Report date:	June 2011
NIWA Project:	GAU11201

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Executive summary

Coral reef systems, seagrasses and mangroves in the tropical Pacific and East Timor region provide a fundamental ecosystem service by moderating the inherent dynamic nature of reeffringed shorelines, and the occurrence or severity of episodic storm or swell-related inundation. Coral reefs and associated carbonate producers such as coralline algae and foraminera also provide the primary sources of carbonate framework and sediments for ongoing reef accretion, and the sediments that form many of the reef islands and shorelines across region.

These coastal ecosystems are expected to be particularly susceptible to the additional cumulative stressors of climate change. The climate changes that are currently anticipated to be experienced this century include significant increases in air and ocean temperatures, changes to mean and extreme rainfall, increase in mean sea levels, and increased acidification of ocean waters. Key issues include:

- Coral reef systems are likely to be most sensitive to increased sea surface temperatures and ocean acidification changes, with interactions between temperature and acidification likely to exacerbate impacts for some species. Significant seasurface temperature-related mass coral bleaching events have occurred in the region over recent decades. Many coral species across the region are already near their thermal tolerance, and bleaching events are predicted to be more frequent in coming decades. Similarly ocean acidity is likely to fall below the level required to sustain coral reef accretion within the first half of this century, assuming carbon dioxide emission rates continue at current rates. Both temperature and acidification impacts are likely to occur earlier in the more coral-diverse western Pacific (Melanesia) than in the central mid-ocean areas. Impacts will include:
 - Significant loss of coral species diversity with a change to more temperature resilient species.
 - Greater expenditure of energy used for coral skeletal growth resulting in changes to coral metabolism, reproduction and larval settlement.
 - Reduction in calcification rate changing the balance between reef-system accretion and erosion. Reduced rate of coral colony growth and reduced skeletal densities, making coral colonies more prone to damage and reduced resilience to recover from disturbance events. Ultimately long-term reductions in ocean pH will lead to internal and external dissolution of calcified reef structures.
 - Changes in the composition of reef flat carbonate sediments to more thermally tolerant producers, such as *Halimeda* spp. and foraminifera.
- Seagrass and mangrove systems are predicted to be most sensitive to increasing sea-level. For both seagrasses and mangroves, their ability to respond to sea-level rise is determined by the ability of species to colonise and extend shoreward, the availability of suitable substrate, and whether sediment accretion balances erosional processes. Seagrass and mangroves on islands with fringing reef environments are

expected to experience the largest impacts of climate change, especially those with low tide ranges, as higher sea levels enable increased wave energy to propagate over reef flats. Seagrass and mangrove systems in some areas may also experience climate change impacts due to increases in rainfall and storm events, High-island environments are predicted to experience impacts from increased land-based sedimentation due to increased rainfall intensity.

At present many coastal ecosystems, particular those close to population centres, are being significantly impacted by increased sediment and nutrient run-off due to catchment and coastal development, lack of sewage treatment, overfishing, and direct impacts from activities such as dredging and marine pollution. Climate change will exacerbate impacts on coastal ecosystems that are already subject to high levels of human impact. Such impacts significantly decrease the resilience of reef systems and their ability to absorb or recover from a disturbance event.

Climate change and on-going human-related impacts on tropical coastal ecosystems in the Pacific and East Timor will impact on two primary mechanisms of coastal protection that regulate physical change in adjacent shoreline position, and are currently provided by these systems:

- Reducing the amount of wave energy dissipation that occurs due to waves breaking on the outer reef crest and further dissipation of wave energy as waves travel over the reef flat and other coastal habitats.
- Changes in the rate and composition of sediment supply, transfers and storage, and losses of sediments within reef flats and adjacent shorelines.

In many coastal margins, natural features or seawalls and urban development prevent shoreward migration of key coastal ecosystems, such that sea-level rise will result in either increasingly narrow distributions (coastal squeeze) or loss of species zonation. Sea-level rise is expected to be particularly problematic where development or infrastructure located in the coastal margins has reduced the area available for seagrass or mangrove retreat. Under sea-level rise rates expected later this century, mangrove stands on atolls and low islands will be most at risk given the low rates of sedimentation and peat accumulation, lack of sediment supply, and typical lack of available space to move landward.

The coastal protection ability of tropical coastal ecosystems will decrease as climate change impacts reduce the distribution and abundance of key coastal ecosystems. Changes in coastal protection function will be site specific, and depend on complex interactions and feedbacks between these physical processes of wave dissipation and sediment supply, coastal morphology, and current and future impacts of humans on these coastal ecosystems.

Irrespective of the future impacts that climate change will have on coastal ecosystems, these ecosystems currently provide coastal-hazard resilience to adjacent human communities. Improving ecosystem resilience in the context of the entire natural coastal-protection system, and maintaining the flexibility to adapt to change are rarely considered, particularly in the context of addressing coastal hazard issues. However, if adaptation for both Pacific societies and the environment are to be effective, recognising these ecosystems as a fundamental component of societal health, and addressing the root causes of many of the human-related

impacts on them are critical. Even with optimal natural coastal protection functions provided by tropical coastal habitats, there are limitations to the protection service. They cannot protect coastal communities from all extreme weather and storm tide events.

As climate change increasingly impacts on coastal ecosystem health, in turn this will most likely gradually reduce resilience and the level of protective functions provided by such ecosystems. Understanding how coastal land-use planning, management and engineering activities and interventions can better add to and complement the protection provided by such ecosystems, rather than detract from it, is fundamental to increasing the resilience of adjacent communities in the long term.

1 Introduction

The coastal ecosystems of island nations of the tropical Pacific and East Timor are expected to be vulnerable to the ongoing impacts of climate change. Climate change is projected to result in significant increases in air and ocean temperatures, changes to mean and extreme rainfall, increase in mean sea levels, and increased acidification of ocean waters (Table 1-1). The characteristics of extreme events may also change; for example, globally the number of tropical cyclone events is anticipated to decrease slightly towards the later part of this century, but the intensity of the most severe events is predicted to increase. Each of these physical changes will drive ecological changes that alter the resilience of coastal ecosystems. The resilience of coastal ecosystems to change will in turn be affected by natural resilience mechanisms, as well as current human pressures that decrease resilience.

In the tropical Pacific and East Timor, coastal ecosystems such as reef, seagrass and mangroves habitats play a fundamental coastal protection role as well as providing many other ecosystem services. This coastal protection function includes:

- 1. Direct protection to the shoreline through reduction in wave energy reaching the shoreline, including direct wave breaking and wave energy dissipation on reef edges, and wave energy dissipation through friction and buffering effects as waves travel over reef flats and seagrass beds and through mangrove stands.
- 2. Carbonate sediment production providing the only or dominant source of sediment input to island shorelines.
- 3. Trapping of sediment, and provision of sediment 'storage' within back reef habitats, mangrove stands and seagrass beds.

Reef, seagrass and mangrove ecosystems, particularly in island settings away from continental margins, are usually critically inter-dependent. From the seaward side, coral reefs provide the buffering and protection from oceanic wave conditions that creates a reef flat or lagoon wave energy climate suitable for mangrove and seagrass habitat, which in turn, when well established, can provide further protection to the shoreline from wave action. From the landward side, mangroves and seagrasses (and other wetland areas on land) filter and buffer coral reef systems from sediments, nutrients and contaminants from land run-off, maintaining a low nutrient environment beneficial for coral reef growth, and reducing the potential for algal-related impacts on coral health.

The purpose of this report is to provide: a) a summary assessment of the potential impacts that climate change may cause on key coastal ecosystems in the Pacific and East Timor; b) how climate change may impact on coastal ecosystem function as a natural coastal protective buffer, and c) the scope for the management of coastal ecosystems to provide an alternative or complementary approach to more traditional forms of coastal risk management as an adaptation strategy.

The summary focuses on particular habitats: 1) coral reefs and associated carbonate producers (focusing on the key contributors to reef building and sediment budgets), 2) seagrasses, and 3) mangroves. Each ecosystem is described in the context of their regional variability and roles in influencing coastal morphology and coastal hazard process, shoreline

change and inundation within different island types and coastal settings. Based on current literature it will assess:

- Their exposure and sensitivity to climate variability and change (and any regional variation in potential impacts) focussing primarily on sea temperatures, ocean acidification, sea-level rise and changes in tropical cyclone activity (Table 1-1).
- Their resilience to climate variability and their capacity to adapt to climate change including an assessment of both natural and human barriers that may impede effective adaptation.
- Ecological resilience and consequential impacts (e.g., macroalgal phase shifts; loss of reef-building corals following hurricane or coral bleaching events).
- Their role as a disaster risk reduction and climate change adaptation option in reducing risks to communities and infrastructure from coastal inundation and erosion.

Climate stressor	Changes and potential projections	Certainty of impact
Increased sea- surface	 Sea-surface temperatures have increased by about 0.7°C since pre-industrial times. 	High, already observed
temperatures	 Projections of average sea-surface warming of 0.5-1.1°C by 2030s and 1.1-3.6°C by 2100 depending on emission scenario. 	
Sea level rise	 Mean sea levels have risen 0.12-0.22 m over the last century with the rate in the Pacific region likely to be close to the global average. 	High, already observed; rate will accelerate over this
	 Plausible increase likely to be between 0.5 – 1 m by 2100 but could be higher. 	century
	 Rises in extreme sea levels are likely to be of a similar magnitude to mean sea-level rise. 	
Ocean acidification	 Global average pH of surface waters has dropped by 0.1 since the mid-nineteenth century to a current value of approximately 8.1. 	High, already observed
	 Projected to decrease by a further 0.1 to 0.3 pH units by the end of this century. 	
	 Atmospheric CO₂ have increased by 40% from a pre-industrial value of around 280 ppm to 387 ppm by the end of 2009. 	
	 Projections for CO₂ of between 500 ppm to 1000 ppm by 2100 depending on emission scenario. 	
Tropical cyclone activity	 No clear consensus whether tropical cyclone activity has changed over recent decades. 	Low for location and frequency; High for
	 Globally, cyclone frequency is likely to decrease or stay unchanged but the intensity of the most severe events is likely to increase. 	increased intensity

Table 1-1: Summary of climate changes and projections in key climate change stressors assumed in this report, based on the 2007 IPCC assessment (IPCC 2007).

2 Coral reefs and associated calcareous systems

2.1 Regional characteristics and variability

2.1.1 Coral and coral reef systems

Coral reefs are shallow marine habitats, common to coastal fringes of land masses in lowlatitude tropical regions. They are biogenic structures (made by marine organisms), taking centuries to millennia to build consisting of a contemporary living veneer of coral and associated reef organisms that overlie past deposits of calcium carbonate (Kench et al. 2009). Complex processes shape the morphology of these solid reef structures including the binding and erosion of carbonate materials. While hard stony (scleractinian) corals are often the dominant architectural forms, other associated carbonate producers, such as coralline algae, foraminifera and the skeletal remains of other organisms, such as molluscs, are also direct contributors to the carbonate components of coral reef systems through:

- Binding and cementing calcareous sediments in the case of crustose coralline algae (Section 2.1.2).
- Carbonate production and a source of sediments that create and maintain landforms associated with reef systems (Section 2.1.3).
- Interactions with fish and invertebrate grazers and bioeroders that play a role in modifying carbonate materials to create or maintain landforms associated with reef systems.

Coral reef environments represent a critical resource upon which all Pacific Island nations depend. Coral reefs provide habitat for significant sources of food, resources for export, and income through tourism. They play an important role in the formation and morphological changes that occur on adjacent shorelines, including providing protection from the full effects of storms and cyclones (Spalding et al. 2001). Globally, and across the Pacific, a suite of human impacts, including overfishing, destructive fishing practices, sediment, and nutrient run-off, tourism, coastal development, navigation infrastructure and climate change threaten coral reefs and the broader ecosystem they support.

Coral reefs habitats are the most diverse of tropical marine ecosystems. Coral diversity is highest in the Indo-West Pacific, centred on southeast Asia (Figure 2-1), with Melanesia incorporating about 14% of the global total of coral reef ecosystems. Different coral species adapted to particular depths and exposures to form somewhat predictable zonation patterns for species found in lagoons, back reefs, reef flats, reef crests and reef slope habitats. The structural diversity of coral reefs is associated with abundant and diverse communities of fish, invertebrates and macroalgae (South and Skelton 2000).



Figure 2-1: Coral species diversity in the Indo-pacific region based on predicted number of species from distributional models. (Adapted from Spalding et al. (2001)).

2.1.2 Crustose calcifying algae

Crustose calcifying algae are found across the entire tropical Pacific (Figure 2-2), and are a key functional group in coral reef ecosystem. Calcifying algae are important both in contributing to coral reef formation, as well as forming crustose coralline algal reefs. Key calcareous algal taxa include both crustose and articulated coralline algae (e.g., *Porolithon, Lithophyllum*) and calcareous green algae (e.g., *Halimeda, Udotea*). They fulfil a number of important ecological roles on coral reefs including contributing to reef calcification and cementation by binding sediments and other carbonate fragments into solid reef structure and reducing reef erosion potential by providing a calcified tissue barrier (Littler 1994). Coralline algae also induce larval settlement of many benthic organisms including coral larvae, and can be a major producer of carbonate sediments (Kuffner et al. 2008).

Calcifying algae are also important in constructing algal ridges at the reef edge in locations with high and persistent (year round) wave exposure, where some coralline algal species (e.g., *Porolithon*) create well developed ridges that provide increased protection to shorelines in their lee by increasing the amount of wave energy dissipated through wave breaking at the edge of the reef (Littler 1994) (e.g., Figure 2-2). Calcareous green algae are more commonly associated with protected shallow areas, though some species (e.g., *Halimeda*) are found on reef slopes (Littler 1994). Calcification in algae gives advantages in tropical reef environments of increased mechanical support to cope with impacts of wave abrasion, as well as reduced impact of herbivory.



Figure 2-2: Reef flat and reef crest crustose coralline algae on Niulakita, Tuvalu.

2.1.3 Carbonate sediments

The primary sources of reef carbonate and the sediments that form reef islands and beaches across the tropical Pacific region are from the calcareous skeletons of coral, calcareous algae (both red algae, discussed above, and green algae, specifically *Halimeda*), foraminifera, and to typically a lesser extent, molluscs, echinoids and other benthic organisms (Kench et al. 2009). Beach and reef sediments are the product of a complex range of biological (e.g., carbonate production, bioerosion and biodissolution) and physical processes (e.g., transport and abrasion, deposition, diagenesis). The long-term balance between the supply and loss of carbonate sediments is fundamental in influencing the net change of both reef systems and associate reef flat islands and shorelines (Kench et al. 2009).

The composition of beach sediments varies widely across both the region and at the local scale, e.g., between lagoon and ocean shorelines (Figure 2-3), and depending on substrate type. Production of carbonate sediments can vary by several orders of magnitude across different reef flat sub-environments, depending on the relative abundance of carbonate-producing taxa (Hart and Kench 2007). In terms of overall carbonate productivity, corals and calcareous algae are generally more important than foraminifera. However, foraminifera tests (Figure 2-4) can contribute much of the sand making up the land area of atolls, for example in Tuvalu, western Kiribati and the Marshall Islands.



Figure 2-3: Beach sediments on the lagoon (left) and ocean (right) coasts of Fongafale Island and Funafuti Island, Tuvalu. The ocean beach sediments are dominated by coral and red algae debris deposited and moved onshore after Cyclone Bebe in 1972, whereas on the lagoon side (and much of the sediment forming the land), large foraminifera tests are the dominant component of the sand and gravels (Collen and Garton 2004)



Figure 2-4: Foraminifera tests from Tuvalu (left) and Halimeda on the reef flat on Kosrae, **Federated States of Micronesia.** Foraminifera photograph courtesy of Prof. John Collen, University of Victoria, Wellington.

Across the Pacific there are differences in dominant foraminifera species. Of the key large foraminifera species contributing to the sand budget on reef flats, *Calcarina* and *Baculogypsina* species tend to occur in the western Pacific and as far east as the Marshall Islands, Phoenix Islands in Kiribati, Tuvalu and Tonga, whereas *Amphistegina* tests are common throughout the Pacific (Yamano et al. 2005). On reef flats foraminifera tend to be found in relatively shallow water depths, often towards the outer edges of reef flats and tend to be epiphytic, preferring a substrate with algal growth, seagrasses or dead coral rubble while they exhibit only limited colonisation in areas of smooth reef flat. Foraminifera tests tend to be relatively more abrasion resistant, for example compared to coralline algae. However, they are sensitive to human impacts such as elevated nutrient levels, and in places where there are large human populations and high nutrient loads to lagoons, such as Tarawa and Majuro, they have largely disappeared (Collen and Garton 2004).

The calcareous green algae Halimeda can also be a significant source of sediment. It occurs on the reef flat (Figure 2-4), but often the most well developed beds occur in deeper water on the outer reef face and fore reef slope. Halimeda are branched and comprise of calcified segments and uncalcified joints. The calcified segments are shed, forming sand, and substantially contributing to critical habitats for other reef organisms (Nelson 2009). Halimeda fragments tend to be weaker, highly friable and tend to break down under transportation and abrasion by wave action.

Exposure and sensitivity to climate variability and change 2.2

2.2.1 Changes in temperature

Coral reefs are highly sensitive to changes in sea temperatures. Thermal related stress is a primary, and most widely reported, factor resulting in coral bleaching, where corals eject their symbiotic algae, leaving the white skeleton visible (Baker et al. 2008, Hoegh-Guldberg et al. 2007, Hughes et al. 2003). Episodes of mass coral bleaching have led to coral mortality, declines in coral cover, shifts in coral species composition, and phase shifts where coral cover has been replaced by macroalgae (e.g., Diaz-Pulido, et al. (2009); Graham, et al. (2007)). Temperature-related bleaching is usually associated with sea temperatures exceeding normal summer maximums by 1-2°C (Eakin et al. 2010).

Isolated and localised bleaching events have been recognised for many decades; but, observations of mass bleaching occurrences, at least of low to moderate severity (Oliver et al. 2009), have significantly increased in frequency over the last 30 years. Significant mass bleaching events occurring in 1982/83, 1987, 1998, 2002, and 2005, (Baker et al. 2008, Hoegh-Guldberg 1999, Stone et al. 1999), with more recent bleaching recorded in Palau in 2010, associated with elevated sea surface temperatures through large parts of Micronesia.

Whilst records of past bleaching events in the Pacific are not as comprehensive as other locations, thermal stress estimates from satellite-derived estimates of temperature variability suggest that 40% of Pacific reefs were subjected to thermal stress that are likely to have resulted in bleaching between 1998 and 2007 (Burke 2011) (Figure 2-5). The 1997-1998 bleaching event was the most globally extensive severe bleaching and coral mortality event on record, with sea-surface temperatures the highest in recorded history. In general, and at a global scale, the risk of bleaching is higher during El Niño periods, which tend to result in higher than normal maximum sea surface temperatures. This is certainly the case in the eastern Pacific but less-so in the western Pacific particularly in the vicinity of the South Pacific Convergence Zone. This region tends to be warmer during La Niña phases, for example mass bleaching was absent during 1997-1998 but occurred in 1998-1999 during the subsequent La Niña phase (Wilkinson 2002).

It is generally accepted that most corals within the tropical regions are living close (within a degree or so) to their thermal limits and are the most sensitive of the key reef system carbonate producers to increased temperature (Hoegh-Guldberg 1999; Hoegh-Guldberg et al. 2007)(Table 2-1). They are extremely sensitive to small increases in sea temperature above summer maximum temperatures with visible number of coral showing signs of bleaching at 1°C above normal, and more extensive bleaching of most coral species

occurring if temperatures of 2°C higher than normal occur for 4 weeks or longer (e.g., Hoegh-Guldberg et al. (2008).



Severe Bleaching
 Moderate or Low Bleaching
 Satellite-detected Severe Thermal Stress

Figure 2-5: Locations of coral bleaching observations and abnormally high sea surface temperatures causing coral reef thermal stress between 1998 and 2007. Figure adapted from Reefs at Risk (Burke 2011) <u>http://www.wri.org/project/reefs-at-risk/</u>.

Since pre-industrial times, sea surface temperatures have increased by approximately 0.7°C with anticipated increases in sea surface temperatures of between 0.5°C to 1.1°C by the 2030s and 1.1°C to 3.6°C by 2100s depending on emission scenario (Donner 2009). Climate models suggest that the temperatures resulting in bleaching events may be exceeded annually within 50 years (Baker et al. 2008, Burke 2011) (**Error! Reference source not found.**). In addition to bleaching, other negative effects of higher temperatures have been recorded on coral metabolism, reproduction and larval settlement (Fitt et al. 2000, Hoegh-Guldberg 1999, Jokiel et al. 2008, Szmant and Gassman 1990).





Temperature will also have an influence on other carbonate producers within coral reef systems. Calcareous coralline algae, foraminifera and Halimeda species are predicted to be able to tolerate higher temperatures than corals (Diaz-Pulido et al. 2007, 2009, Fuentes et al. 2010) (Table 2-1). Temperature increase is anticipated to slightly increase calcification rates for calcareous coralline algae, with changes in seasonality, growth and reproduction possible, but temperature-linked mortality unlikely. However, projected sea-surface temperatures this century may exceed optimal temperatures for photosynthesis, growth and reproduction (Diaz-Pulido et al. 2007). Sea-surface temperature increase is also expected to be linked to higher microbial growth rates and potential incidence of calcareous coralline algae diseases, though disease susceptibility across the Pacific region is poorly known.

Organism	Thresholds	Key factors determining sensitivity and impacts
Reef building coral	Upper threshold of around 30°C for most coral species 1°C to 2°C above current summer maximum temperatures	Coral species and diversity: A bleaching event does not necessarily mean that all coral species on a particular reef system will bleach, with variability of tolerance of thermal stress between species well recognised. Platy or branching corals, such as <i>Acropora</i> are usually amongst the first to bleach, whereas massive coral forms, such as <i>Porites</i> , tend to appear more robust to bleaching and can tolerate slightly higher temperatures, albeit still generally within the 1-2°C range (Brown 1997, Done and Jones 2006, Hoegh-Guldberg 1999).
		Habitat: Variability in thermal tolerance also occurs among corals due to habitat, with coral species found in intertidal reef pools and back reefs where water temperatures can be significantly elevated. However, relative increases in water temperatures in these environments are likely to place the same levels of stress on coral species (Hoegh-Guldberg et al. 2008).
		Duration of thermal stress. Variable but in general one month of sea surface temperatures at 1°C above
		Ocean acidification: Potential for increased ocean acidification to increase the sensitivity and decrease thermal tolerance of some species.
Coralline algae	Upper thresholds likely to be around 33°C to 35°C	Species and habitat: Intertidal species tend to be more temperature tolerant that sub-tidal species.
	for tropical species	Duration of thermal stress: Some indication that many species unable to survive permanently at 35°C.
Foraminifera	Upper thresholds likely to be around 34°C with some species up to 39°C	Species: Larger, carbonate producing foraminifera likely to tolerate potential sea-surface temperatures projected over this next century and may well become dominant reef builders and sediment producers(Fuentes et al. 2010)
Halimeda	Likely to have a high tolerance to sea-surface temperature increase.	Thermal tolerance: Some indication that <i>Halimeda</i> species can tolerate long-term exposure to high sea-surface temperatures.

Table 2-1:	Key factors that influence the sensitivity and impact of sea surface temperature on
coral reefs	and associated carbonate systems.

The variability of thermal stress and resistance to bleaching across different coral species is well established (e.g., Hoegh-Guldberg & Salvat (1995)). Local or regional variation in thermal tolerance within a coral species has also been observed, implying site-specific adaptations to thermal stress (Berkelmans 2002, McClanahan et al. 2004). Particular oceanographic, biological or environmental factors can also create areas where thermal stresses are reduced or avoided, for example areas of ocean upwelling, areas of fast flowing currents, and reef areas that are shaded, either directly by the surrounding reef structure or in locations where there tends to be increased cloud cover (Woodridge and Done 2004). For example reef flat corals are more commonly exposed to higher temperatures and will have evolved higher thermal resistance with temperature thresholds close to the average upper temperature thresholds of their locality (Grimsditch and Salm 2006). However, actual genetic resistance is generally unknown, as is any clear evidence of the capacity for evolutionary adaptation or acclimatisation to warmer temperatures (Baker et al. 2008, Hoegh-Guldberg 1999). As this acclimatisation has occurred over a much longer period than the rates of temperature change anticipated due to global warming, and as sea temperatures in these warmer environments will also increase, these corals are likely to experience similar temperature-related stresses (Hoegh-Guldberg et al. 2008).

Symbiotic dinoflagellates also differ in their heat tolerance, suggesting that thermal stress resilience requires tolerance of both the coral host and their symbiont (Hoegh-Guldberg 1999, Ulstrup et al. 2006). It has also been suggested that coral bleaching could be an evolutionary mechanism (known as the adaptive bleaching hypothesis) to adapt to warmer sea conditions where less temperature resistant zooanthellae are replaced with more resistant strains (Baker 2004, Buddemeier and Fautin 1993) but this remains debatable (e.g., Obura (2005)) with uncertainty as to the potential for coral acclimatisation. In the absence of a physiological adaptive response to the rates of temperature rise projected to be experienced this century, it is fairly certain that there will be a change in the relative composition and abundance of species, resulting in less diverse and resilient reef ecosystems with reef communities, as we presently know them, eventually disappearing if sea surface temperatures continue to rise rapidly (Hoegh-Guldberg 2009a).

2.2.2 Ocean acidification

Approximately 25% of carbon dioxide currently emitted from anthropogenic sources enters the ocean resulting in changes in ocean chemistry. Increasing atmospheric concentration of CO_2 leads to more dissolution of CO_2 in surface waters, resulting in increased acidity via the formation of carbonic acid (Department of Climate Change and Energy Efficiency 2011). The reduction in the carbonate ion concentration associated with increased acidity also leads to a reduction in calcium carbonate saturation state (e.g., Hoegh-Gulberg et al. 2007). Since preindustrial times global seawater pH has decreased on average by 0.1 units (8.16 to 8.05) and is projected to decrease by a further 0.1 to 0.3 pH units by the end of the century (IPCC 2007). Acidification levels are less in tropical oceans than in higher latitude regions due to higher sea surface temperatures resulting in lower dissolution rates of atmospheric CO_2 . Carbonate ion concentration have also reduced by approximately 15% (222 to 186 micromol per kg) and of aragonite supersaturation from 4.6 to 4.0 (Guinotte and Fabry, 2008).

Organism	Thresholds	Key impacts and sensitivities
Reef building coral	No net carbonate accretion on coral reefs when aragonite saturation is less than 3.3 and carbonate ion reduces below 200 µmol kg ⁻¹ Thresholds where coral reefs shift from net accretion to erosion will vary greatly from reef to reef (Kleypas and Yates 2009). Thresholds likely to be reached with atmospheric carbon dioxide concentrations of approximately 480 ppm with almost all reefs in a state of dissolution with concentrations of 560 ppm (Silverman et al. 2009) pH of around 7.7 likely threshold for reef development to cease (Fabricius et al. 2011).	 The balance between accretion and erosional processes will change with reduced calcification rates, which will vary between species of somewhere between 10-60% for a doubling of pre-industrial carbon dioxide concentrations Reduced calcification rates will lead to (Hoegh-Guldberg et al. 2007): Reduced linear extension rate of coral colonies Reducing skeletal density, (to maintain physical extension or growth rates), leading to decreased resilience to, and recovery, from other stressors such as storm damage, and potentially increased erosion from carbonate grazers selecting lower-density substrates. Maintenance of skeletal growth and density through greater expenditure of energy at the expense of other activities, such as reproduction which could reduce larval output and reduce recovery from disturbance events. Permanent under-saturation of aragonite saturation state will lead to internal and external dissolution of calcified structures (Silverman et al. 2009)
Coralline algae	Thresholds are predicted to be lower for coralline algae than for corals due to the higher sensitivity of magnesium calcification to increased ocean acidification.	Crustose coralline algae have been shown to be particularly sensitive due to high magnesium calcite being more soluble than aragonite at similar carbon dioxide concentrations. Few studies in tropical areas but mesocosm experiments provide indications of a 40% reduction in growth rates, 78% decrease in recruitment, and 92% reduction in total area covered by crustose coralline algae (e.g., Kuffner et al. (2008)). The sensitivity of coralline algae may compound impacts on corals, as coralline algae are key settlement substrates for corals (Hoegh-Guldberg et al. 2007, Nelson 2009).
Foraminifera	Thresholds unknown, but are likely less sensitive than corals.	Varies between and within species but decrease in shell mass generally expected for most foraminifera. Some suggestions that net calcification of <i>Baculogypsina</i> and <i>Calcarina</i> spp. could increase slightly under intermediate level of carbon dioxide concentrations (Fujita et al. 2011).
Halimeda	Thresholds unknown, but are likely less sensitive than corals.	Limited information but increases in carbon dioxide concentrations likely to reduce ability to calcify (Kleypas and Yates 2009)

Table 2-2:	(ey thresholds, impacts and sensitivities of ocean acidification on coral reefs and	ł
associated	arbonate systems.	

As most reef building and reef sediment producing organisms form biogenic calcium carbonate, they are sensitive to a reduction in carbonate ions, which essentially makes it more energy intensive or difficult to form biogenic calcium carbonate. However, the specific mechanisms relating changes in carbon dioxide in seawater to calcification rates are complex and still poorly understood with the calcification processes likely to differ between species, species life stages and at different stages of calcification (Kleypas and Yates 2009).

Calcite and aragonite are the two main crystalline forms of calcium carbonate produced. Calcite is the principal form deposited by foraminifera and coralline red algae (high magnesium calcite) with aragonite formed by most scleractinian corals and calcifying green algae (*Halimeda*) with calcification rates correlated best with aragonite saturation state for these latter organisms (e.g., Langdon, 2002).

Despite a large number of laboratory and mesocosm studies demonstrating the sensitivity of marine calcifying organisms to the amount of carbon dioxide in seawater and resulting changes in seawater chemistry (e.g., Guinotte & Fabry, 2008), to date there have been limited field observations, though see for example (Fabricius et al. 2011). Declines in calcification rate and skeletal density of *Porites* have been observed at monitored locations on the Great Barrier Reef indicating a decrease of 21% between 1988 and 2004, or an approximate decline of 1.29% per year in calcification rates (Cooper et al. 2008). Whilst it is not possible to wholly attribute these changes to ocean acidification the magnitude of response is consistent with the reported changes in seawater chemistry.

Aragonite saturation levels above 4 are determined to be optimal for reef calcification and accretion and below 3.3 will result in reef erosion (Guinotte et al. 2003). Current global models that project aragonite saturation horizon in the ocean (Figure 2-7) indicate that it will continue to shallow significantly in the next century due to ocean acidification (Cao and Caldeira 2008, Guinotte et al. 2003) with levels likely to fall below those required to sustain coral reef accretion by the 2050s (Carpenter et al. 2008, Guinotte et al. 2006, Hoegh-Guldberg et al. 2007). Across the Pacific region changes in ocean acidity and carbonate saturation will be variable with areas in Melanesia, specifically the Coral Sea and East Timorese region, already experiencing marginal aragonite saturation levels at current carbon dioxide levels (Figure 2-7). In general if carbon dioxide levels were stabilised at present day levels, changes in coral reef systems would still occur but they would remain coral dominated and accrete carbonate in most areas of the Pacific (Hoegh-Guldberg et al. 2007). If carbon dioxide levels continue to increase at current rates, by the decade between 2020-2029 aragonite saturation levels across the entire Pacific region will be below optimum levels and by 2069 only a few areas focussed on Polynesia, and parts of Micronesia will remain at adequate levels (Guinotte et al. 2003).

Resistance to ocean acidification will vary from species to species and with location. Whilst calcification will again increase if acidification is reversed, such a reversal would appear unlikely at least this century. At present there is very limited information on whether coral and associated carbonate species can adapt to the potential increases in ocean acidification expected this century and beyond (Kleypas and Yates 2009). Recent field observations in PNG (Fabricius et al. 2011) close to volcanic carbon dioxide seeps suggested that between pH values of 8.1 (present day) to 7.8 (expected this century) the following could occur:

- Reductions in coral diversity, recruitment and abundances of structurally complex reef framework occurred, as well as shifts in competitive interactions between taxa.
- Coral cover remained constant with a loss of fast-growing, structurally complex corals and a progressive shift to slow-growing, long-lived massive *Porites* corals despite low rates of calcification.



Figure 2-7: Changes in aragonite saturation predicted to occur as atmospheric carbon dioxide concentrations (ppm) increase. Spatial variations in aragonite saturation are shown from preindustrial concentrations of carbon dioxide (280 ppm), through present day (380 ppm) to projected values for 2100 (650 ppm). Source: Hoegh-Guldberg et al. (2009b).

Below a pH of 7.7, reef development ceased. The field observations show that whilst reef systems may persist, together with increased temperature stresses, it is with the likelihood of much reduced species diversity, structural complexity and resilience (Fabricius et al. 2011). Mass extinction events at major geological intervals in the past could be linked with increased ocean acidification, assuming estimated levels of atmospheric CO₂ resulted in similar changes in pH to those likely to occur over this century, though the data gaps in the

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Elevating ocean CO_2 concentrations by protonating HCO_3 also potentially allows aquatic primary producers to more than double their rates of photosynthesis (Nelson 2009). This change in photosynthetic rate has the potential to modify the relative abundance of calcifying versus fleshy algae, as well as potential changes in the competitive dominance of macroalgae over invertebrates, mediating further bottom-up changes in trophic relationships and community dynamics.

At a localised level, photosynthesis within dense seagrass beds is known to increase pH levels within and in surrounding seawaters, and can stimulate calcification of calcareous organisms within the seagrass beds (Semesi et al. 2009). Highly productive dense and shallow seagrass beds could provide some localised mitigation against the effects of ocean acidification on calcification rates of calcareous algaes, foraminiferas and other sediment producing organisms living with the seagrass beds, and for nearby reef top corals. The buffering capacity of carbonate sediments associated with reef ecosystems as it relates to acidification impacts is also poorly understood.

The ecological ramifications of changes in species diversity are unknown, but are ultimately important in understanding what coral reef systems will look like in a higher atmospheric CO_2 environment. As such, maintaining the adaptive capacity of existing ecosystems through is crucially important to maintaining the resilience of these systems in the face of climate change and the coastal protection role and other ecosystem services that they provide.

2.2.3 Sea-level rise

The present day form of coral reef systems throughout the tropical Pacific have evolved in response to sea level changes over the Holocene (Kench et al. 2009). Over this time reef system growth has responded to sea-level rise in a number of different ways, at the most generic level defined as 'keep-up', 'catch-up' and 'give-up' reef systems (e.g., (Neumann 1985, Woodroffe 2002).

Over the 20^{th} century global mean sea levels have increased by on average 0.17 m ±0.05 m (i.e., 1.7 mm/yr ±0.5 mm/yr) (IPCC 2007) with rates over the tropical Pacific region similar to the global mean. There is still much uncertainty as to the potential magnitude of sea-level rise that will be reached by the end of the century and beyond, much of which relates to the ensuing response of the large polar ice sheets. The IPCC Fourth Assessment Report concluding a range of sea-level rise of 0.18 to 0.79 m by the 2090s but were careful to note that higher values could not be excluded (IPCC 2007). Whilst potential rises of up to 2 m by 2100 have been suggested a more plausible current estimate suggests a rise of around 0.5 to 1.0 m by 2100 compared to 2000. (Department of Climate Change and Energy Efficiency 2011).

The growth of coral reef species, if healthy and actively growing, are expected to be able to accommodate the rates of sea level rise expected this century (Burke 2011). Indo-Pacific coral reefs have recorded vertical growth rates of up to 30 mm/yr but more typically up to 10 to 12 mm/yr (e.g., Kench, et al. 2009) which are within the range of predicted sea level rise. Rates will vary depending on location within a reef system resulting in different coral growth

potential. Sea-level rise could potentially see the re-establishment of reef growth through submergence of present-day partly-emerged reef platforms where coral reef and coralline algae growth is constrained by low tide exposure (Spencer 1995).

There is much less information available on the potential impacts of sea-level rise on the other reef carbonate producers. Whilst rates of crustose coralline algae growth are typically much slower than coral species, sea-level rise over this century is unlikely to cause major changes in distribution. Some depth–related habitat losses may be experienced but equally likely is the potential for increases in available substrate for colonisation. Likewise sea-level rise is unlikely to directly impact foraminifera and *Halimeda* carbonate production with impacts likely to be indirectly related to the loss or creation of suitable substrate habitat. Potential examples include loss of habitat due to erosion of seagrass beds due to increased wave energy caused by increased water levels over a reef flat, or increased habitat due to increased amount of coral rubble over a reef flat caused by increased wave breakage of weaker coral skeletons.

Actual net rates of reef system accretion are typically much lower than coral growth rates, being the net result of coral reef and coralline algae growth, carbonate sediment production, and bio-erosion processes. Based on Holocene rates of reef growth, which show considerable variability, the threshold for long-term sea-level rise beyond which reef systems are unlikely to maintain vertical parity is likely to be around 8-10 mm/yr (Spencer 1995). This rate is also critically dependent on the other climate and human-related stressors on coral reefs which add considerable uncertainty as to how present-day reef systems may respond. Whilst such rates of sea-level rise rates are not likely to result in catastrophic drowning of reef systems this century (Spencer 1995).

2.2.4 Cyclone intensity and frequency

Coral reef ecosystems across the tropical Pacific region between 7° to 25° north and south of the equator are episodically impacted by tropical cyclone events (known as typhoons in the north-west Pacific). On average the South Pacific region experiences 9 cyclone events of Category 1 or greater per year compared with the North-West Pacific which has an annual average of 26. Year to year variability in cyclone characteristics appear in both regions primarily due to El Niño Southern Oscillation. There is still uncertainty as to whether tropical cyclone activity has changed over the last few decades as any changes are small relative to this natural variability.

Cyclone and typhoon events commonly result in damage to reef structure, both through direct wave action but also due to dislodgement and movement of coral, which can result in further coral damage at lower depths. Given the limited number of cyclone events in any one season, particularly in the South Pacific, and the relatively narrow width over which damaging conditions typically occur (10's of km), the likelihood of a particular reef being impacted in any one season is low. Resulting reef damage and disturbance can occur in a number of forms. Disturbance can be highly variable even over short spatial distances or depths, particularly for cyclone winds less than about 150 km/hr (Kench et al. 2009). Disturbance depends on the trajectory of the cyclone and its characteristics, particularly the intensity, in relation to the reef system. Disturbance varies with the structural form of coral

species, with compact forms usually more resilient to storm damage than erect, branching forms (Clark and Morton 1999, Connell et al. 1997). Disturbance also varies with the length of time since the last damaging cyclone event occurred, and the intensity of prior events. Winds of over 200 km/hr can have much more significant impact on the reef system as a whole, reducing it to largely rubble, making it more difficult for coral re-establishment and reef recovery, the impacts of which can last decades (Scoffin, 1985).

Reefs that are already weakened by other threats, such as acidification, are more likely to be damaged by cyclones (Burke 2011). Severe cyclone events are the only mechanisms where large amounts of coarse coral-derived sediment can be delivered to reef platforms and can provide an important sediment supply mechanism for the ongoing development of reef top islets and island shorelines.

Crustose coralline algae are less sensitive to cyclones and tropical storms due to having generally smaller size and crustose forms and well developed holdfasts decreasing likelihood of dislodgement, as well as their greater resistance to impacts of sandblasting from resuspension of sediments. Cyclone events may also have a positive effect on crustose coralline algae by increasing substrate availability by damaging and removing competitors such as turf and fleshy algae.

Globally it is anticipated that the frequency of cyclone events may decrease but that the intensity of the most severe category 4 and 5 events could increase. In the Southern Hemisphere whether there will be an increase in intensity is presently indeterminate, with more indication of a potential increase in the north-west Pacific:

- Any decrease in the frequency of events, particularly category 2 or greater, would reduce the potential incidence of reef damage for any particular reef location and may increase the amount of recovery time available between cyclone events. However, such decrease may impact on the supply of coral rubble sediments to the reef flat and adjacent shorelines.
- Category 4 of 5 cyclone events are relatively rare. Any increase in intensity of a particular event would likely increase the spatial extent and level of damage compared to a present day event, and would likely have a more significant impact on overall reef damage and ability to recover (than changes in frequencies of less severe events). Little information is available but it has been suggested for the Great Barrier Reef than an increase in half a cyclone category could result in 50-60% greater loss of coral cover (Fabricius et al. 2008). The increase extent and severity of damage would also result in a longer recovery time with potential increased negative impacts on maintaining the ecosystem function of the reef.
- Irrespective of any changes in cyclone frequency and intensity, the effects of temperature and acidification stressors on structural densities of corals will make them more susceptible to storm-related damage and will slow coral recovery after an event. In the short-term this may increase the supply of coralrubble sediment to reef flats and adjacent shorelines.

For islands close to the equatorial margins of cyclone occurrence, for example the southern Marshall Islands and eastern islands in the Federated States of Micronesia in the northern hemisphere, and East Timor, Tuvalu, Tokelau and eastern Polynesian islands in the southern hemisphere, cyclone occurrence is closely linked to ENSO state, particularly the occurrence of El Niño conditions. Whilst there is presently no consistent indication of discernible future changes in ENSO amplitude or frequency, any anthropogenic-related changes in ENSO characteristics may be most noticeable in tropical cyclone activity at these margins of cyclone occurrence.

2.2.5 Other potential stressors

Climate change will particularly exacerbate impacts on reefs that are already subject to high levels of human impacts, caused by activities such as coastal development, overfishing and destructive fishing methods, watershed and marine-based pollution, nutrient loading, coastal infrastructure such as navigational channels, and sedimentation impacts (Burke 2011, Goldberg 2008, Hughes and Connell 1999). Currently the Pacific is one of the least threatened regions with around 50% of reefs classified as threatened and about 20% at high or very high risk with most of the threatened reefs associated with islands with larger populations (Burke 2011), although specifically threatened locations are apparent (Figure 2-8). However, declines in coral reef cover have occurred across the Indo-Pacific with long term losses of 1% over the two recent decades, and an increased rate of annual loss due to combined stressors of 0.72% per year from 1997 to 2004 (Bruno and Selig 2007). Stressors include:

- Crown of Thorns starfish (Acanthaster planci) epidemics have been observed since the 1960s across the Indo-Pacific region. These starfish feed only on live corals, and plagues have resulted in up to 95% mortality of coral cover. Their cause is unknown but associated with severe storms and high rates of terrestrial nutrient runoff (Sebens 1994). Reefs in American Samoa suffered an outbreak in 1978 but have since recovered with other outbreaks recorded in Micronesia, the Tuamotus and Society Islands of French Polynesia since the 1970s and 1980s.
- Disease in scleractinian corals was first reported in the 1970s, thus far occurring in 106 species including soft corals on reefs in 54 countries (Spalding et al. 2001). While diseases are more prevalent in the Caribbean, and also associated with reefs subject to other human threats, diseases are becoming more common on coral reefs of the Pacific (Burke 2011). There is also strong evidence that reefs are more vulnerable to epidemics following coral bleaching events, suggesting that increasing temperatures from global climate change will make coral reefs more susceptible to disease epidemics (Done and Jones 2006).
- Runoff from land: In addition to sedimentation associated with runoff from storm and cyclone event, corals are sensitive to human activities on land including: 1) runoff from coastal development and watershed activities which can impact corals through decreasing water clarity, and in some cases by smothering corals; 2) Elevated nutrient inputs to coastal areas, for example

from catchment and coastal margin land-use practices and inadequate sanitation infrastructure and sewage disposal, which can lead to excessive macroalgae growth and algal blooms. This can result in a range of impacts from reduced water column clarity to complete phase shifts in ecological states within the reef system (Hughes et al. 1999, McCook 1999); and 3) pollution from opencut mining which is a significant threat in Papua New Guinea (copper and gold mining) and in New Caledonia (nickel) (Burke 2011).

Overfishing and destructive fishing: Overfishing and destructive fishing has increased substantially in the last decade in the Pacific due to coastal population growth (Burke 2011, Zann 1994), with most fishing impacts centred upon population centres. However, industrial scale fishing, primarily by international fleets, and targeted on larger predatory fish, also has top-down impacts on coral reefs. Aquaria trade also reduces target species, often removing pre-reproductive individuals. Fishing of key grazers results in increased algal growth, also can also result in increased abundance of bioeroding species due to release from competition or predation (McClanahan and Muthiga 1988, Mumby et al. 2006). Bleached reefs or those damaged by a cyclone event are particularly susceptible to algal over-growth and phase shifts, where key grazers have been over-fished and are less likely to recover. Destructive fishing techniques such as dynamite and cyanide fishing also reduce resilience of coral reefs system to other disturbances.

Both existing anthropogenic impacts on reefs and increasing climate change stresses, particularly relating to temperature and ocean acidification, decrease a reef system's resilience, defined as the ability of the reef to absorb or recover from a disturbance or change event (Grimsditch and Salm 2006). Such disturbance events can be episodic, such as a severe cyclone, medium-term such as over-fishing impacts or increasing nutrient run-off, or longer-term such as changes in oceanographic conditions due to climate change, or a combination of stressors. For example corals bleach at lower temperatures in waters with a lower pH with such interactions, although still poorly understood, likely to hasten the deterioration of reefs. Significantly these disturbances can initiate changes in the ecological state and function of a reef system, specifically a phase shift (e.g., Done 1992) from one that is dominated by coral species to typically one that is macroalgal dominated.

Once such a phase shift occurs it can be difficult to reverse, typically requiring cessation of the key stressors though even this may not return systems to their prior state (Hughes et al. 2010). For example, on Moorea, between 1991 and 2006, coral reefs were impacted by a cyclone and four bleaching events. The bleaching event in 1991 resulted in a loss of coral cover of between 22-51% and subsequent colonisation by turf algae with coverage between 16-49%. However, over the next decade this phase shift did not remain with coral cover returning to pre-disturbance levels but with a different community composition, with increased cover of *Porites* and full return of *Acropora* but a reduction in a number of other species (Adjeroud et al. 2009).



Figure 2-8: Risks to coral reef classified by present day local threats and how this risk may change due to climate change. The top map shows reefs at risk classified by integrated presentday threats including coastal development, overfishing and destructive fishing, watershed sediment, nutrient and pollution run-off and marine-based pollution. The middle and bottom maps show the risk classification where present-day local threats are combined with projections of thermal stress and ocean acidification for 2030 and 2050 respectively (Burke 2011).

2.3 Impact on their coastal protection function

Climate change related impacts on coral reef and associated reef carbonate systems will impact on the coastal protection function of these systems in two main ways:

 reducing the amount of wave energy dissipation that occurs due to wave breaking on the outer reef crest and potential wave energy dissipation as waves translate over the reef flat changes in the sediment supply, and sediment characteristics, to adjacent shorelines.

It is these changes in wave energy reaching island shorelines and / or changes in reef flat and shoreline sediment budgets that will be the primary driver of physical change in adjacent shoreline position.

Coral reefs and reef flats dissipate much of the ocean wave energy protecting both island and mainland shorelines. Across broad reefs this may be up to as much as 99% but more commonly around 80% of the wave energy (Sheppard et al. 2005). Any increase in water depth over the outer reef crest or over the reef flat, or changing reef flat surface characteristics, will potentially increase the magnitude of wave energy reaching any adjacent shoreline. Such changes will be highly site specific and also dependent on tide range characteristics and existing reef flat topography, for example the occurrence of conglomerate platforms.

Sheppard et al. (2005) assessed the reduction in coastal protection caused by a mass coral bleaching in 1998 in the Seychelles that resulted in the loss of living *Acropora* thickets and massive *Porites* corals over the fringing reef crests and seaward portions of the reef flats. This resulted in three observed changes:

- Removal of the coral skeleton, as it was eroded by wave action, increased the depth of water over the reef flat by the same amount as the reduction in elevation of the coral surface.
- Reduction in wave friction dissipation as the structure of the reef flat surface changed from being highly irregular due to the coral thickets and boulders to a much smoother surface.
- Loss of the reef crest corals slightly changed the characteristics of the wave breaking zone seaward of the reef crest.

Averaged across 14 monitored locations, these changes combined to result in an average of a 35% increase in wave energy reaching the shoreline relative to the situation prior to the bleaching event.

With the rates of sea-level rise likely to be experienced this century a critical issue is whether rates of reef crest and reef flat growth will be able to track closely to sea-level rises ("keep-up"); or whether, due to the effects of other stressors, there will be a lag in response ("catch-up"); or reduction in vertical accretion (for example, where long-term phase sifts occur from coral to algal dominance, or from reef-framework building species to non-framework species). To date present-day vertical accretion in general have been close to sea-level rates over the last century (Spencer 1995) but if reef crests, in particular, are not able to track closely to the accelerated sea-level changes expected, even small changes in the difference between sea-levels and reef crest levels will have important implications for increases in wave energy across reef flats.

Increased wave energy over reef flats, changes in reef crest and reef flat coral compositions, and ecological phase shifts all will impact on the rates and relative composition of reef

carbonate sediments that contribute to either reef structure or as sediment input to associated reef islands and shorelines (Kench et al. 2009), for example:

- Increased wave energy over reef flats has the potential to impact on reef flat substrate cover such as coral rubble and seagrass beds which is important habitat for foraminifera, calcareous algae and other carbonate reef biota.
- Loss of coral structural complexity at the reef crest or over shallow-water reef flats, such as thickets of branching corals such as *Acropora*, will reduce the supply of coarse coral skeletal fragments to the sediment budget.
- Shifts in ecological state will have a significant influence on the rate of sediment supply and the relative composition of reef carbonate sediments (Kench et al. 2009) with a reduction in coral-derived sediment and a relative increase in other carbonate sources (e.g., calcareous algae, foraminifera), the relative proportions of which will depend on the particular local environment. Such shifts can result in changes to the overall carbonate budget.

3 Seagrasses

3.1 Regional characteristics and variability

Seagrass meadows are important shallow marine ecosystems in the tropical Pacific and East Timor, providing important food sources, habitat, breeding and nursery areas for a wide variety of marine species. They play a role in providing some coastal protection, in trapping and storage of reef flat sediments, and in carbon sequestration through conversion of dissolved carbon dioxide in seawater to oxygen and the transport of carbon to deep water.

Diversity of species is highest in the west of the region, toward the global peak of seagrass biodiversity over insular Southeast Asia (Spalding et al. 2003). Highest diversity of 12 or more co-occurring species has been recorded in Papua New Guinea and Vanuatu, decreasing to one species in French Polynesia (Figure 3-1) and no species in a few remote atoll nations, such as Tokelau. The region is dominated by tropical seagrass taxa, with key genera including *Cymodocea, Enhalus, Halophila, Halodule, Syringodium, Thalassia*, and *Thalassodendron* (Spalding et al. 2003). *Thalassia hemprichii* is the dominant seagrass in Micronesia and Melanesia and forms dense meadows on reef platforms, as well as in shallow muddy sediments (Coles et al. 2003).



Figure 3-1: Seagrass species diversity in the Pacific Islands. Note data was not available for East Timor, but is likely similar to Papua New Guinea.

Seagrass ecosystems in Micronesia, Melanesia and Polynesia are generally found on fringing reefs, on the protected side of barrier reefs and back reefs, and on the larger high islands in estuary systems (Coles et al. 2003) (Figure 3-2). Seagrass can be found on rocky and reef substrates or in lagoons or estuaries with fine sand or mud (Coles et al. 2003).

Seagrass meadows are generally nutrient-limited, and may be associated with populated areas with higher nutrient availability (Coles et al. 2003, Waycott et al. 2007). For example the expansion of seagrass along the southern part of Tarawa Lagoon in Kiribati has coincided with population increase (Figure 3-2). However, negative impacts of human development on seagrass are common, with mechanisms including increased turbidity and sediment deposition due to land-based impacts.



Figure 3-2: Seagrass beds on a fringing reef on Kosrae, Federated States of Micronesia (left) and in sheltered lagoon location on Tarawa, Kiribati (right).

Seagrass ecosystems in the Pacific and East Timor are usually associated with high productivity and standing stock of organic matter, and associated with numerous other species including epiphytic algae, small epifauna and infauna, larger crustaceans, gastropods, juvenile and adult fish, and marine megafauna including the green turtle and dugong (Coles et al. 2003), Key ecosystem roles include primary production, creation of above- and below-ground habitat structure, provision of settlement substrate, nutrient and contaminant filtration, sediment filtration and trapping, oxygen production, nutrient regeneration and recycling, organic matter accumulation, and carbon sequestration (Orth et al. 2006, Short et al. 2000). Their physical role in coastal protection includes wave and current energy dampening, reduction in turbidity and stabilisation of sediments (Hemminga and Duarte 2000). Economic valuation of seagrass ecosystems based primarily on their role in nutrient cycling suggests a global value of the ecosystem service at US\$19,004 per hectare per year (Costanza et al. 1997).

Historical declines in seagrass cover have occurred over much of the region often due to infrastructure and tourism development, population growth and catchment land use changes, though historical information is rarely available to quantify these changes (Coles et al. 2003, Coles 1996, Maragos 1993).

3.2 Exposure and sensitivity to climate variability and change

3.2.1 Changes in temperature

Seagrasses will in general experience thermal stress from projected increased water temperatures, and desiccation stress from overexposure to warmer air temperatures expected over this century. In the tropical Pacific region this is most likely to impact on inter-tidal and shallow sub-tidal seagrasses.

In general tropical seagrass species are likely to be more tolerant to metabolic changes than temperate seagrasses. Impacts will be species dependent with most tropical Pacific species tolerant of short-term warmer air and water temperatures (e.g., *Halophila ovalis* and *Thalassia hemprichii*), while temperate species (e.g., *Syringodium isoetifolium* and *Cymodocea serrulata*) are less tolerant and restricted to deeper waters (Bridges and McMillan 1986, McMillian 1984, Waycott et al. 2007).

Indirectly, warmer air temperatures are also likely to result in osmotic impacts due to increased evaporation (Short and Neckles 1999). Increasing temperatures are also likely to result in changes in seagrass metabolism through increased rates of both leaf respiration and photosynthesis (Short and Neckles 1999). The ratio between photosynthesis and respiration is expected to decrease as respiration rate increases faster than photosynthetic rate as temperature increases (Waycott et al. 2007), though the effect of higher atmospheric CO₂ levels on this ratio is unknown. There is some suggestion that the productivity of tropical seagrasses is likely to begin to decline at seawater temperatures above 30°C (Fong and Harwell 1994) and that seagrasses at the upper end of their thermal tolerance will be significantly impacted by a 2°C increase in temperature (Ralph 1997) though information on the response to sustained high temperature exposure is limited for most Pacific species (Waycott et al. 2007).

Over inter-tidal and shallow reef areas elevated temperatures, particularly during low tides around midday, can exceed 40° and could result in increased potential for seagrass loss if these short-term temperatures increase, and it is suggested that photosynthetic processes can become significantly affected at temperatures of between 38-42°C (Campbell et al. 2006). Loss of species and/or shifts in species composition due to higher temperatures in the region are likely only to occur where species are growing close to their physiological limits, such as in tidally restricted areas (Bjork et al. 2008), or they are subject to multiple stressors. Whilst temperature is also likely to shift distribution of seagrass species this is more likely to be an issue for temperate species or species at the boundaries between tropical and temperate areas, rather than within the tropical Pacific region.

3.2.2 Increasing carbon dioxide concentrations and ocean acidification

Seagrasses tend to be carbon limited and to date no identified change has been observed in seagrass distribution or productivity due to increased carbon dioxide concentrations or due to the reduction in seawater pH (Waycott et al. 2007). Whilst it is expected that seagrasses are likely to respond positively to increased CO₂ enrichment with increases in photosynthesis and growth, long-term exposure experiments have been less conclusive (Short and Neckles 1999). To an extent this will depend on seagrass species and the particular mechanisms for inorganic carbon uptake. Those species that favour direct uptake of carbon dioxide may have a slight advantage over those that utilise bicarbonate, with the latter route favoured by some tropical seagrasses (Waycott et al. 2007). However, any effect on species composition will be minimal as other influences will dominate. The buffering capacity of carbonate sediments associated with seagrass ecosystems as it relates to acidification impacts is poorly understood.

Increasing ocean acidification over the reduction in pH projected this century is unlikely to limit photosynthesis or counteract the effects of increasing carbon dioxide (Beer et al. 2006).

In field observations at the locations of volcanic seeps in Papua New Guinea (where pH varied from 8.1 to 7.8 comparable to the reduction expected over this century) seagrass communities in the areas with lower pH and higher carbon dioxide concentrations had three to four times the shoot densities and below ground biomass but reduced seagrass species diversity compared to areas with higher pH (Fabricius et al. 2011). Dense seagrass meadows have also been shown to cause diurnal pH fluctuations due to photosynthesis, resulting in raised local pH values particularly during low tides or in areas with slow water movements (Semesi et al. 2009). It is suggested that shallow water, dense and highly productive seagrass meadows could counter the effects of ocean acidification in localised areas and provide a buffer to calcifying organisms that occur within the seagrass beds or closeby.

3.2.3 Sea-level rise

There are no known locations within the Pacific and East Timor regions where sea-level rise over this last century has directly impacted seagrass meadows. However, increased sea level rise, over the rates likely to be experienced this century will result in increased water depths resulting in less than ideal growing conditions (Green and Short 2003, Short and Neckles 1999). This is most likely to impact on changing the location of the maximum and optimal depth limits of each species based on photosynthetic compensation depths, resulting in changing spatial distributions of seagrass as increased depths reduce available light (Short and Neckles 1999). In the tropical Pacific region this is most likely to affect deep water seagrass species and in locations where turbidity levels influence seagrass occurrence.

In some locations seagrasses may be able to move landwards to compensate, for example the sub-tidal seagrass beds found along the outer edge of the lagoon sandflats in South Tarawa, Kiribati. However, in many locations, for example fringing reef environments, mangrove stands, beaches or more significantly coastal defences or urban development may impede the ability to move landward (coastal squeeze) resulting in increasingly narrow distributions and potential loss. Sea-level rise may also influence tidal currents and flushing characteristics, particularly in shallow water lagoon environments. Any effects will be extremely site specific and could include increased turbidity, uprooting and erosion of substrate, and seagrass recruitment. However, increased flushing and tidal exchange may also be beneficial in reducing the elevated water temperatures that can occur in shallow water environments and associated temperature-related stresses.

Indirect effects of sea-level rise may have a more significant impact on seagrass meadows. Where seagrasses occur on fringing reef flat environments, increased water depths over the reef flats will potentially result in increased wave energy propagating over seagrass beds. This has the potential to cause erosion of the seaward edge of the beds (Figure 3-3) or result in increased potential of uprooting of seagrasses and loss of unconsolidated substrate.

3.2.4 Cyclone intensity and frequency

Storm and cyclone events are capable of dislodging surface biomass, with larger events capable of uprooting substantial areas of below ground biomass in seagrass meadows (Green and Short 2003, Short and Neckles 1999). Intertidal and shallow subtidal seagrass meadows are most susceptible to storm disturbances.
Storm and cyclone events typically result in increased erosion, suspension and redeposition of sediments resulting in increased turbidity, decreased water quality and potential smothering of seagrass beds. This can occur due to erosion and suspension of shoreline or nearshore or reef sediments, or due to increased rainfall intensity that typically occurs during such events resulting in increased catchment or coastal margin run-off, sediment loads, nutrients and pollutants. Effects can be highly localised.

Storm disturbance can result in changes to community structure in diverse seagrass meadows by favouring early colonising species at the expense of later colonising seagrass species (Short and Neckles 1999).

Given the episodic nature of such events and the changes in tropical cyclone activity anticipated over this century, direct changes or additional impacts on seagrass systems due to changes in tropical cyclone activity may be indeterminate for many parts of the region (but would most likely be most noticeable over areas at most frequent cyclone risk such as the Western Pacific north of the equator and the area around the Solomon Islands, Vanuatu and Fiji in the South Pacific). Rather increased impacts on seagrasses are potentially likely to occur due to:

- Any increases in water levels over shallow water areas and fringing reef flats due to sea-level rise enabling more wave energy reaching the seagrass beds during cyclone events. This will depend critically on how the adjacent coral reef systems respond to sea-level rise.
- Increased rainfall intensity rates during such events, influencing turbidity, sedimentation, and nutrient supply. Conversely many seagrass taxa appear to be tolerant of changes in water salinity due to heavy rainfall, for example, tolerance of salinities of 2 ppt in Yap, Micronesia (Bridges and McMillan 1986).

3.2.5 Other potential stressors

Other direct and indirect stressors may also be associated with climate change. Changes in cloudiness and hence solar radiation may affect seagrass species currently at their visible light limits. Shallow water seagrasses have also been shown to be sensitive to UV-B radiation due to reduced ozone levels (Dawson and Dennison 1996, Green and Short 2003, Waycott et al. 2007). Seagrass disease epidemics have also been linked to both temperature increases and elevated salinity but it is not known whether this has been an issue in the Pacific region (Short and Neckles 1999).

Tropical seagrass beds are known to vary seasonally and from year to year due to natural events such as floods or cyclones. For example, seagrass beds on Suva Reef in Fiji show an oscillation in abundance regressing in some years and extending towards the lagoon in other and have been attributed to high turbidity and siltation due to nearby foreshore reclamation and construction activities (McKenzie and Yoshida 2007). To date it is human activities that are having the most significant impacts on seagrass beds throughout the region, albeit impacts tend to be on a fairly localised scale, with existing seagrass monitoring efforts suggesting that seagrass bed are in a fair condition where they are monitored in the region (McKenzie et al. 2006). Key stressors include:

- Losses through dredging or reclamation, for example the construction of the runway on the reef flat at Yela on Kosrae which resulted in both direct loss due to the reclamation but also loss of surrounding seagrasses due to high turbidity and sedimentation from the construction (Maragos 1993).
- Increased turbidity and sedimentation caused by: 1) increased runoff and sediment loads from land due to catchment land use changes, development activities and clearing of natural vegetation, or removal or decreased effectiveness of coastal margin sediment traps and buffers such as coastal wetlands and mangrove stands, or 2) due to reduced water circulation, for example due to causeway construction.
- Increased nutrient loads from agriculture activities or inadequate sanitation systems which can lead to increased competition from macroalgae, increasing epiphyte loads, or increased turbidity from phytoplankton blooms (Houk and Camacho 2010).

3.3 Impact on their coastal protection function

Most predicted impacts of climate change on seagrass habitats, in conjunction with existing human-related stressors, suggest declines in seagrass cover in the Pacific and East Timor. As seagrasses perform a number of functional roles including sediment stabilisation, filtering of terrestrial sediment inputs, and attenuation of flows, this has the potential to decrease coastal protection.

Wave heights reaching the shoreline are highly sensitive to reef flat levels and characteristics. Whilst seagrass influence on reef coastal food webs, nutrient budgets and carbon sequestration are well recognised, their influence on the dynamic regime is less well established. However, seagrass beds can play a contributing role to the coastal protection function of a coastal ecosystem, most significantly where they occur on fringing reef or shallow lagoon environments (water depths less than a few metres). They play two primarily roles. Firstly they can dissipate wave energy propagating over reef flats, reducing the amount of wave energy reaching mangrove stands or the shoreline, and secondly they help bind marine and terrestrial sediments and reduce erosion and turbidity through trapping sediment and managing the interchange of sediment between the shoreline, reef flat and the ocean. They contribute to a cumulative effect on wave attenuation as part of the coastal reef ecosystem and are dependent on substantial wave energy reduction due to wave breaking along the outer edge of a fringing or barrier reef system.

Seagrass beds dissipate wave energy by trapping sediment and raising local substrate levels (Figure 3-3), thereby decreasing water depth, and depth-limited wave heights, over the reef flat, and through wave interaction with the seagrass canopy which reduces wave orbital velocities through drag effects as waves propagate over the bed. However, seagrass beds need to be well established, with sparse or patchy beds contributing little in reducing reef flat wave conditions propagating over them. Chen et al. (2007) concluded that bed density needed to reach a threshold of 1000 shoots per square metre to contribute to wave attenuation. The degree of wave attenuation can also vary seasonally (Chen et al. 2007) and with latitude. Above-ground biomass (and attenuation effectiveness) is lowest in the tropics in

summer between 0° to 10° (Duarte and Chiscano 1999) with attenuation effectiveness tending to be at its maximum when plants are reproducing.

The most significant factor influencing wave attenuation over a seagrass bed is the depth of water with attenuation highest when the seagrass canopy occupies greater than 50% of the water column (e.g., Fonseca & Cahalan, 1992; Koch et al. 2006; Ward et al. 1984). This suggests that seagrass would be less effective in dissipating waves at high tide, particularly in locations with high tide ranges, and during storm conditions when water levels over the reef flat tend to be higher. However, the degree of attenuation also depends on the period of the wave conditions propagating over the reef flat and seagrass can still providing some attenuation under such conditions (Koch et al. 2006).

Where seagrass beds presently play a coastal protection role, in general climate change and sea-level rise can be expected to reduce the influence that they have on wave attenuation through either loss or change in seagrass habitat or through reduced influence of the seagrass canopy through increased general water depth. Such changes will be highly site specific and difficult to predict (Short and Neckles 1999) given the dynamic interannual nature of change that can occur in seagrass meadows, and the multiple stressors at different temporal and spatial scales that typically drive seagrass change (Orth et al. 2006). However, where seagrass provides a coastal protection role on reef flat and shallow fringing lagoons, the ongoing effectiveness of this role will depend significantly on:

- The ability to maintain seagrass meadows of a sufficient density to maintain both the sediment substrate (which reduces water depth), and the influence of the seagrass canopy on wave attenuation.
- Maintenance of the width of the seagrass bed. (Chen et al. 2007) demonstrated that increased wave attenuation occurred with increasing bed width in the direction of wave propagation.
- The vertical response of the outer reef edge to sea-level rise in minimising an increase in wave energy on to reef flats or shallow lagoon areas. Whilst seagrasses can be resilient to wave action, losses can occur when wave or storm action erodes the substrate, particularly at the seaward edges (Figure 3-3) or results in excessive levels of sediment transport and sedimentation which either smothers existing beds or limits seed establishment (Koch et al. 2006).
- The long-term ability to trap and bind sufficient sediment within the substrate to maintain vertical accretion of the substrate at a similar rate long-term sea-level rise.



Figure 3-3: Eroding seaward edge of a seagrass bed on the north coast of Kosrae, Federated States of Micronesia.

4 Mangroves

4.1 Regional characteristics and variability

Mangrove ecosystems in Micronesia, Melanesia and Polynesia are generally found in soft sediments in inter-tidal zones protected from wave action including protected estuaries and deltas, lagoons, and fringing coastal habitats with low wave energy (Spalding et al. 2010). Mangroves have physiological adaptations to allow them to adapt to variable salinities and tidal inundation. The largest mangrove stands are found on the larger, high islands in Melanesia (Papua New Guinea, Solomon Islands, Vanuatu, Fiji, New Caledonia) due to a higher sediment supply, though forests are also found on smaller islands with sufficient rainfall and sediment supply (Spalding et al. 2010).

Different mangrove species have distinct niche preferences, often showing strong spatial patterns of zonation associated with salinity, elevation (inundation), and sediment properties (Spalding et al. 2010) (Figure 4-1). Vertical zonation patterns are most common with dominant species on seaward shores (e.g., *Rhizophora stylosa, R. samoensis* and *R. x selala*) changing to species such as *Brugiera gymnorhiza, Heritiera littoralis* and *Xylocarpus* sp. on the landward side (Figure 4-1). Canopy heights are typically highest inland, reaching as high as 30 m, for example, in Kosrae and Pohnpei in the Federated States of Micronesia where disturbance to canopies is rare (Spalding et al. 2010). Temporal patterns of zonation representing successional dynamics are also observed (Spalding et al. 2010).



Figure 4-1: Mangrove zonation from mean tide to high tide typical of Papua New Guinea and the Solomon Islands. Based on Ellison (1997).

Mangroves occur in a variety of tropical coastal settings (Figure 4-2), (e.g., Kjerfve (1990)) including:

- Deltaic mangrove forests occurring at the mouths of large river systems, found on larger volcanic islands such as Fiji, (e.g., at the mouths of the Nadi and Rewa rivers) and Papua New Guinea. In Papua New Guinea deltaic floodplains
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accounts for more than 50% of the southern coastline on the Papua New Guinea mainland and about 10% of the northern coastline (Sullivan 1990). These typically have the most developed patterns of mangrove species zonation (Figure 4-1).

- Estuary and estuary lagoon systems occurring on high islands, for example Kosrae in the Federated States of Micronesia where species zonation includes *Rhizophora mucronata* found close to river mouths, *R. apiculata, Bruguiera gymnorhiza* and *Sonneratia Alba* in the interior of the mangrove and patches of *Nypa fruticans* and *Xylocarpus granatum* towards the interior (Whitesell et al. 1986).
- Lagoon systems on low islands with no river inputs, for example Fangakakau Lagoon on Tongatapu, Tonga.
- Fringing mangrove systems along atoll lagoon margins and tidal passages (e.g., Tarawa Lagoon), sheltered areas on fringing reef systems (e.g., Kosrae, Federated States of Micronesia)) and back reefs behind barrier reef systems, e.g., Pohnpei, Federated States of Micronesia).

On high islands, rainfall has a major influence of mangrove characteristics, with climatic variation across some of the larger islands influencing mangrove distribution and ecology (Watling 1985). For example, across Vitu Levu in Fiji, the drier leeward north-west shoreline, where rainfall amounts are lower and seasonal, strand widths tend to be narrower, have less species diversity, with stunted mangrove growth, larger strand gaps and areas of salt flat, relative to the windward south-east shoreline, where mangrove growth is more productive and diverse, due to the higher, and near-continuous year-round rainfall (Ellison 2009).

Mangrove ecosystems are highly productive, and maintain high standing stocks of biomass similar to terrestrial tropical forests (Spalding et al. 2010). They are associated with numerous other species including epiphytic algae, small epifauna and infauna, larger crustaceans, gastropods, and juvenile and adult fish (Spalding et al. 2010). Key ecological roles include primary production and export of carbon and nutrients through the detrital food web, creation of habitat structure, and provision of settlement substrate. Other important roles include nutrient and contaminant filtration, sediment filtration and trapping, and organic matter accumulation (Spalding et al. 2010).

Their physical role in coastal protection includes shore protection through attenuation of waves and swell (including tsunami waves) resulting in a reduction of wave run-up and potential inundation, and reduced erosion of shoreline sediments via wave energy dissipation and sediment trapping (Alongi 2008). Economic valuation of mangrove ecosystems based primarily on their role in coastal protection suggests a global value of US\$3679 per hectare per year with additional benefits for mangrove-fishery linkages and forest resource use (Sathirathai and Barbier 2001).

Diversity of species is highest in Papua New Guinea, closest to the global peak of mangrove biodiversity in the Indo-West Pacific (Spalding et al. 2010). The region is dominated by tropical mangrove taxa, but also includes species with broad distributions such as *Avicennia marina* (Spalding et al. 2010). Highest diversity of over 40 co-occurring species has been

recorded in Papua New Guinea, with rapid decrease in the number of mangrove species in an easterly direction across the Pacific Islands (Figure 4-3).



Figure 4-2: Examples of mangrove environments in the tropical Pacific. A: Extensive mangrove strand on the north-west coast of Viti Levu, Fiji, B: Lagoon mangroves behind the fringing coastal berm on the south coast of Kosrae, FSM, and C: Fangakakau Lagoon on Tongatapu, Tonga; Fringing mangroves stands in lagoon tidal passage in Tarawa (D), on a sheltered open coast fringing reef on Kosrae (E) and on back reefs behind the barrier reef system on Pohnpei (F).



Figure 4-3: Mangrove species diversity in the tropical Pacific region. (Spalding et al. 2010).

4.2 Exposure and sensitivity to climate variability and change

4.2.1 Changes in temperature

Mangrove distribution in the tropical Pacific is not limited by temperature, and distribution is unlikely to be significantly influenced by the increases in sea surface temperatures experienced over this next century. Mangroves have been shown to be able to survive in higher sea temperatures than is currently projected over this next century, but with clear declines in photosynthetic rate above seawater temperatures of 33°C (Alongi 2008). Changes in community composition and diversity are more likely, as more thermally tolerant species may out-compete less tolerant species (Alongi 2008).

The primary effect of increasing temperature on tropical mangroves is on productivity but still uncertainty remains as to what the effect may be within the Pacific region. Both photosynthesis and respiration are limited by maximum mid-day leaf temperatures, with increasing temperatures potentially reducing mangrove production through increased cost of respiration, particularly where there is a decline in humidity and rainfall (Lovelock and Ellison 2007). However, over much of the tropical Pacific, climate change is likely to mean changes in rainfall amounts, frequency, seasonality, duration or intensity, and their interaction with increasing temperature may affect individual species' responses. With increasing levels of atmospheric CO₂ (see next section) vegetative growth, particularly where growth is currently limited (e.g., the southern latitudes of Fiji, and litter production may well increase (Ellison 2009).

Increasing temperature is also likely to result in changes in reproduction, with expectations of changes in times of flowering and fruiting. Local topography and meteorological conditions are expected to influence sensitivity to increased temperature, for example changes in salinity or relative humidity due to local conditions can ameliorate impacts of increases in aridity caused by increasing temperature (Done and Jones 2006).

4.2.2 Increasing atmospheric carbon dioxide

Increasing CO_2 is expected to increase productivity through accelerated rates of photosynthesis as for most plants (Lovelock and Ellison 2007). For example, *Rhizophora mangle*, showed increased biomass, branching, leaf area, and maturation and reproduction with doubling of CO_2 concentrations, though these changes were also associated with decreases in plant nutritional quality (Farnsworth et al. 1996).

However, other environmental factors have been shown to determine growth rate response under increased CO_2 concentrations, with CO_2 having little influence where growth rate was limited due to salinity and conversely increased growth by up to 40% with increased (twice ambient) CO_2 when growth was limited by humidity (Ball et al. 1997). As such, it is possible that there will be little or no change in mangrove production with increasing CO_2 , though community composition is likely to change due to species-specific responses to climate change (Alongi 2008). Advances in the time of flowering due to rising atmospheric CO_2 are also predicted to occur, with the potential for desynchronisation with key pollinators (He and Bazzaz 2003).

4.2.3 Sea-level rise

Mangrove response to recent sea-level rise has showed varied responses, with some locations where sediment accretion has kept up with sea-level rise (Alongi 2008). Historically, mangroves were unable to sustain sea level rise of >1.4mm y⁻¹ (Ellison 1993). Current evidence suggests that at many locations shoreline migration is occurring and mangroves have been keeping up with sea level rise over the last century where sea-level rise rates have been on average 1.7mm y⁻¹ (Alongi 2008). However, even with positive increments in surface accretion due to the sediment trapping function of mangrove ecosystems, many mangrove ecosystems show a relative loss of elevation due to sub-surface subsidence (Cahoon et al. 2006, Gilman et al. 2008, Krauss et al. 2003).

Regardless, future accelerated sea-level rise is likely to have the most significant influence of all the climate-change-related impacts on mangroves. Basic estimates suggest that a reduction of about 13% in mangrove area could occur directly due to sea-level rise in the 16 Pacific Island countries where mangroves are indigenous with an assumed sea-level rise of 0.88 m between 1990 and 2100, with estimates shown for each country in Figure 4-4 (Gilman et al. 2006).



Figure 4-4: Approximate estimates of change in mangrove area (%) in response to relative sea level change for 16 Pacific Island countries. The sea-level rise assumed was based on the upper projection of the IPCC Third Assessment Report of 0.88 m between 1990 and 2100 (Gilman et al. 2006).

The local effect of sea-level rise on mangroves will be a product of both local and regional factors and is likely to be temporally and spatially variable (Figure 4-5), but this will depend on a range of environmental conditions. Mangrove response to changes in incident physical processes tends to be gradual, resulting in changes in their extent, structure and species composition (Ellison 2009). The ability to respond to sea-level rise is primarily determined both by the ability of mangrove species to colonise and extend shoreward, tolerance to flooding, and the availability of suitable substrate and sediment accretion (Alongi 2008).

In many coastal margins, natural features or structures and seawalls prevent shoreward migration of mangroves (Figure 4-5 and Figure 4-6), such that sea-level rise will result in either increasingly narrow distributions or loss of species zonation (Gilman et al. 2008). Sea-level rise is expected to be particularly problematic where coastal defences, development or infrastructure located in the coastal margins results in coastal squeeze reducing the area available for mangrove retreat (Ellison 2004).

Mangrove capacity to advance with rising sea level is also dependent on whether or not rates of production and trapping and retention rates of sediment substrate exceed rates of erosion (Done and Jones 2006). Apart from some information from Pohnpei, Kosrae and American Samoa, little information is currently available across the Pacific on mangrove sedimentation rates and how sea level is changing relative to mangrove surfaces (Gillman et al. 2006). Lower rates of relative sea level rise (< 1 mm y⁻¹) are expected to be sustainable for most species, assuming adequate space for shoreward migration. However, higher rates more typical of likely future relative sea-level rise rates over this coming century (~5-10 mm y⁻¹)

would require high input of terrestrial sediments to maintain a comparable rate of accretion. For high islands, as high sedimentation rates are associated with higher rainfall, runoff, and nutrient loading, other disturbances may also influence the sustainability of mangrove capacity for sea-level rise (Done and Jones 2006). Under sea-level rise rates expected later this century, mangrove stands on atolls and low islands will be most at risk given the low rates of sedimentation and peat accumulation, lack of sediment supply, and typical lack of available space to move landward.

In existing mangrove stands increased frequency or duration of inundation will potentially result in increased stress where mangrove zonation occurs based on inundation preferences (Ellison 2009). This can reduce photosynthesis and productivity, oxygen uptake and potential mortality, where the lenticels of aerial roots become inundated for excessive periods.



Figure 4-5: Generalised scenarios for mangrove response to relative sea-level changes. (Gilman et al. 2006).

Table 4-1: Factors influencing the vulnerability of mangrove systems to change under sealevel rise. Adapted from McLeod and Salm (2006).

Least vulnerable	Mostvulnerable
High tide range <	• Low tide range
Vertical substrate accretion rates similar to long-term sea level rise rates	Vertical substrate accretion rates lower than sea-level rise rates
Mangroves in deep sediments on high islands	Atoll and low relief carbonate islands
Areas tectonically stable	Areas subsiding due to tectonic movements or sedimentary loading on deltas
Mangroves backed by low-	Mangroves blocked by high topography, seawalls or other development
Mangroves in remote locations	Mangroves close to large populations
Rivers -	• No rivers
Mangroves surrounded by extensive dense mangrove forests	 Fringing and isolated mangrove strands
Little change in hydrology or inter-tidal salinity	Changes in drainage patterns or inter-tidal salinity



Figure 4-6: Landward transgression of mangrove constrained by seawalls at Nukunuku, Tongatapu (left) and by natural features at Hieghéne, New Caledonia (right).

4.2.4 Cyclone intensity and frequency

Tropical cyclones and large storms can impact through direct destruction of mangroves and in the most severe cyclones mass mortality of the mangrove ecosystem. The seaward-most mangroves often experience a zone of high wave impacts that are more likely to sustain damage, while buffering impacts on inshore mangroves. Typically this zone tends to be relatively narrow, but in the more severe events the damage zone can reach interior mangroves and result in slow or no recovery. Winds and extreme rainfall result in defoliation over a zone close to the cyclone path (Jaffar 1992). Such events typically also cause high rainfall, which on high islands, can result in high run-off of freshwater and sediments. Large sedimentation deposition events can be particularly damaging through smothering of mangrove systems (Ellison 1999). More detailed information on specific impacts and recovery of mangroves due to cyclone events in the Pacific region is lacking.

While mangrove ecosystems are known for their ability to provide coastal protection through sheltering from wind, storm surge and waves, changes in tropical cyclone activity may well reduce the resilience of these ecosystems to storm damage by changing the return frequency of large storms, with less time between events to recover (Done and Jones 2006). However, given the changes in tropical cyclone activity anticipated over this century, direct changes or additional impacts on mangrove systems due to changes in tropical cyclone activity would most likely be noticeable over areas at most frequent cyclone risk (the Western Pacific north of the equator and the area around the Solomon Islands, Vanuatu and Fiji in the South Pacific).

Increased impacts on mangroves are potentially likely to occur due to:

- Any increases in water levels over reef flats due to sea-level rise enabling more wave energy to reach the mangrove stands during cyclone events. This will depend critically on how the adjacent coral reef systems respond to sea-level rise.
- Increased rainfall intensity rates, which are projected to increase of the order of +20% within 100km of the tropical cyclone centre, influencing sedimentation, salinity and nutrient supply, and in conjunction with strong winds the zone of defoliation. While sediment inputs are necessary for mangrove shoreward migration, the associated pollutant inputs can ameliorate or exacerbate impacts of climate change.

4.2.5 Other stressors

In addition to rising mean sea level and increased levels and frequency of extreme high waters, several other forces can affect mangrove distribution, structure and health. These include clearing mangrove vegetation; reclaiming areas of mangrove for construction or aquaculture activities; changing sediment budgets from land development activities or construction of seawalls or reclamations; displacing native species with alien invasive species; vegetation insect infestations, fungal flora pathogens, and other diseases (Donnelly and Bertness 2001, Ellison 1993, Ellison 1996, 1999, Gilman 1999, Saintilan and Wilton 2001).

Many of these pressures can also reduce mangrove resistance and resilience to the additional stress associated with relative sea-level rise and climate change. Furthermore as mangroves are functionally linked to neighbouring coastal ecosystems, including seagrass beds, coral reefs, and coastal margin and catchment habitats, degradation of these ecosystems due to climate change impacts will also impact on mangrove systems.

4.3 Impact on their coastal protection function

Mangrove forests dissipate wave and tidal current energy through bottom friction and, more significantly, by drag friction due to above ground root, pneumatophore, trunk and branch density (e.g., Massel et al. (1999); Quartel et al. (2007)). This latter effect can reduce wave heights in mangroves by as much as 7.5 times than by bare bottom friction alone (Quartel et al. 2007). Mangroves also indirectly affect the propagation of waves by trapping sediment and organic matter, raising bed levels, further limiting shallow water wave heights.

The effectiveness of mangroves in providing shoreline protection is highly context specific, depending on the geomorphology of the area and the frequency and magnitude of storm events that have the potential to cause shoreline change (Ewel et al. 1998). Lacambra et al. (2008) summarise other key conditions and attributes that influence the degree of coastal protection provided by mangroves as:

- Width of the mangrove forest: This depends on species and density and tide range with suggestions, mainly from Asia, of between 100 m and 1500 m as being optimum. In a Pacific setting with limited wave action such as fringing mangroves within lagoon systems, even narrow widths of mangroves will dampen waves, help trap sediments and provide some stabilisation to adjacent shorelines. Alternatively in deltaic settings, where there is often less well developed fringing or barrier reefs to provide initial wave breaking, wave damping may occur over a much larger width, although this depends on the structure and density of the mangrove.
- Mangrove density and strand structure: The age and size of trees within the strand, forest structure, species types, composition and distribution all influence wave damping through friction effects due to above ground root structure, trunks and low branches and foliage. Different species have been observed to generate different drag forces (Quartel et al. 2007), for example in Asia observations of wave heights of 1 m being reduced to 0.3 m within a 50 m strand width of *Avicennia*, and 50% reduction in 100 m stands of *Sonneratia* (Lacambra 2008), 75% reduction through 200 m mangrove width (Massel et al. 1999). In some cases higher water levels can result in higher wave damping where the wave interacts with a dense structure of low-lying branches and foliage.

Where mangroves do provide a coast protection function the effects of climate change on this function will be highly site specific. It is likely that in many situations, sea-level rise will result in increased levels of wave energy reaching the seaward edges of mangrove stands. However, the amount of wave energy reaching the shoreline will depend on:

- a) how increased water level and wave conditions (or other ongoing anthropogenic stressors) affect the width of mangrove strand (e.g., through erosion of the seaward mangroves, or potential landward migration)
- b) the density and structural characteristics (for example loss of zonal species adapted to particular tidal heights or exposures, increased water depths over surface root systems), and
- c) the specific vertical substrate changes within the mangrove system.

Whilst some of the largest shifts in mangrove strand extent and structural characteristics may occur in deltaic and estuarine systems, particularly in locations prone to cyclone events, it is likely along areas with narrower stands and limited sediment inputs, such as fringing stands within atoll lagoon systems where the impacts on coastal protection function are likely to be greatest.

5 Potential for adaptation

5.1 Diversity of function and tolerance

Diverse coastal ecosystems (i.e., seagrass meadows, mangrove forests, diverse coral reefs) have been linked with a higher capacity to adapt to climate change due to the broader functional roles and tolerances of multiple species (Duffy 2006). On an individual species basis, species can be identified by their degree of tolerance to stressors such as thermal stress (Bridges and McMillan 1986), and sites can be identified as to their sensitivity to particular stressors or multiple stressors based on the presence/absence or proportion of tolerant species. Niche partitioning (e.g., intertidal zonation/exposure zonation) of species can be used to predict movement of species distributions with changes in sea level and exposure to higher air temperatures at low tide, and those species that are unlikely to successfully adapt to climate change.

5.2 Shoreward expansion

In many coastal margins, natural features or coastal defences or urban development prevent shoreward migration of coastal habitats, such that sea-level rise will result in either increasingly narrow distributions (coastal squeeze) or loss of species zonation (Gilman et al. 2008). Sea-level rise is expected to be particularly problematic where development or infrastructure located in the coastal margins has reduced the area available for coastal habitat retreat (Ellison 2004). For example, seagrasses or mangroves in some locations may be able to shift shoreward onto areas inundated by sea-level rise, but coastal development is expected to impede shoreward movement on most developed shorelines. Lack of suitable sediments and occupation by competing species (e.g., mangroves in the case of seagrass) can also limit shoreward expansion.

5.3 Resilience and recovery

Adaptive capacity is a system's concept related to the dynamic response of an ecological community to disturbance and stress. This is influenced by both the diversity-driven response and the response of key species. Disturbance due to events can be species specific, with variability in species' tolerances of localised impacts, such as high temperature events, sediment or nutrient loads, salinity changes, increased ocean acidity, and storm events. Coral species are known to vary in the thermal tolerance, primarily due to varying thermal tolerance of their algal symbionts (Hoegh-Guldberg 1999, Ulstrup et al. 2006). Seagrass species are also known to vary in their thermal tolerance (Bridges and McMillan 1986). Different carbonate phyla on coral reefs are expected to vary in their tolerance of ocean acidification due to differences in carbon chemistry making with some species (e.g., coralline algae) less susceptible to impacts of acidification (Fabricius et al. 2011). Species also vary in their tolerance of storm damage, with structural form of coral species having a large influence on their resilience to storm damage, and compact forms usually more resilient to storm damage than erect, branching forms (Clark and Morton 1999, Connell et al. 1997).

Adaptive capacity also includes the ability to quickly recover or recolonise a disturbed area. For example, some seagrass species have the capacity to recover or recolonise via both reproduction and seed dispersal, and dispersal and successful colonisation of vegetative fragments. Examples of seagrass species with rapid recruitment include *Halophila*,

Halodule, Zostera while slower colonisers are *Thallassia, Cymodocea* (Waycott et al. 2007). Coral species also vary in recruitment potential, with some species known for fast, opportunistic colonisation of newly disturbed habitats, while others are known to be slow colonisers (Connell et al. 1997).

5.4 Linkages between coastal ecosystems

Coastal ecosystems are inter-dependent, such that impacts to one ecosystem can have follow-on impacts on neighbouring ecosystems. Coral reefs provide buffers from storm waves, such that less wave energy is propagated to inner coastal ecosystems such as mangrove forests and seagrass meadows. Mangrove and seagrass ecosystems are less likely to suffer storm damage if there is an intact reef system nearby, benefiting from the coastal protection role provided by the coral reef. Coral reef health has been similarly linked to presence of mangrove forest and seagrass meadows. Mangroves and seagrasses filter out land-based sediments and utilise land-based nutrients, limiting the dispersal of sediments and nutrients to the clear oligotrophic waters of the neighbouring coral reef. Without this ecosystem service, coral reefs might suffer from excessive sedimentation which decreases reef health and reduces juvenile recruitment. Eutrophication can also occur without this filtering service, and excess nutrients promote algal growth at the expense of coral species in reef communities. As such, maintenance of large areas of intact coastal ecosystems will have greater capacity to buffer the impacts of climate change.

6 The role of coastal ecosystems in coastal hazard adaptation

6.1 Introduction

Coral reef systems, seagrasses and mangroves in the tropical Pacific and East Timor region provide a fundamental ecosystem service by regulating the natural dynamics of reef-fringed shorelines, and dampening the severity of episodic storm or swell-related inundation. This is primarily done through reducing offshore wave energy reaching the shoreline, and in controlling inputs, transfers and storage, and losses of carbonate and other coastal sediments.

The effects of climate change, (as well as on-going human impacts), will degrade this regulating function and the dynamic equilibrium of reef-fringed shorelines. Effects are most likely to be felt first (and possibly most significantly) through changes in the impacts of episodic extreme climate and weather events (such as cyclones acting upon higher sealevels).

In terms of coastal erosion and inundation hazards, adaptation options for communities and their assets are often categorised under three basic options: planned retreat, accommodation or protection. Pacific Island countries, and indeed all countries, have no option but to adapt to the effects of climate change. Hard-engineered defences aimed at reducing the potential effect of coastal hazards are typically used and cited by atoll (and many other coastal) communities as the socially-preferred option of reducing such risks.

6.2 Impacts of hard defenses on the protection function of ecosystems

Seawalls and other engineered coast protection measures are, in most cases, not as permanent as communities 'protected' by them assume, particularly in atoll settings where the range of construction options and materials available are limited. In many situations the construction of such defenses and the perception of the protection that such structures provide often results in:

- Other detrimental or root causes of human impacts on ecosystem functioning being ignored.
- Increasing intensification of development or urbanisation in what is still a hazard-prone area.

Consequently human-related coastal hazard problems and impacts associated with hard protection measures become more complex over time.

The immediate negative environmental impacts of typical forms of linear hard defenses in exacerbating erosion in front of and on adjacent sections of coast are well known. However, where such structures essentially become permanent features there will be longer-term impacts that will constrain the natural adaptive capacity of reef-related ecosystems and shorelines to adjust and respond to the long-term effects of climate change, most significantly:

- Where habitats and ecosystems respond to sea-level rise or long-term increase in wave energy by migrating landward (such as mangrove strands, seagrass beds and beaches), fixed defenses or urbanisation limit the natural ecosystem response to rising sea levels. These habitats are constrained in a "coastal squeeze" between the defenses or urbanization on the landward side and the pressures directed landwards associated with climate change at their seaward edge. This limits their flexibility to adapt and to sustain the functions, including protection services, that such ecosystems provide.
- Where linear defenses also limit episodic storm overtopping or overwashing of the coastal berm or motu and resulting reduction in deposition of sand and gravel on to the land behind the shoreline, it can limit the gradual build-up of land levels.

6.3 Managing for resilience

Improving ecosystem resilience and maintaining the flexibility to adapt to change in the light of increasing knowledge (adaptive management) are rarely considered, particularly in the context of addressing coastal hazard issues, but are critical in the context of long-term sustainable risk reduction and adaptation within Pacific Island coastal communities.

Coastal hazard risk reduction and adaptation activities need to be carried out in the wider context of managing for reef ecosystem resilience. The regulating services provided by reef systems, mangroves and seagrasses act as a natural absorption system. Other coastal-related risk-reduction and adaptation activities need to recognise and enhance, not impact on, this natural function to increase the resilience of adjacent communities (e.g., ProAct Network (2008)).

Ongoing adaptation for both society and the environment in the Pacific region will be most effective where maintaining or maximising ecosystem services underpin the building of resilience to climate and non-climate risks (e.g., Colls et al. 2009). Most significantly this will enable improved capacity to:

- Manage the uncertainties associated with future climate change.
- Incorporate more effective land use planning as a cost-effective risk-reduction and adaptation strategy.
- Address many of the current underpinning human-related stresses on coastal ecosystems that reduce the level of service they provide, limit their ability to absorb and recover from episodic disturbances, and naturally adapt to climate change pressures.

Lessons learned and key components of reef system resilience and ecosystem-based adaptation approaches are captured in a range of guidance publications (for example Colls et al. (2009); Grimsditch, Salm (2006); Bjork et al. (2008); McLeod and Salm (2006)).

It is well documented that coral reef, seagrass and mangrove ecosystems that are subject to present-day human-related stressors are more likely to succumb to the additional stressors placed on them due to climate change, leading to longer recovery times and increased

likelihood of phase shifts, loss of diversity, and resulting reduction in ecosystem services. Avoiding or reducing non-climate-related ecosystem degradation by focusing on minimising human-related stresses, such as excessive nutrient and sediment runoff and overfishing, is a key aspect both in increasing the ability for these ecosystems to absorb and recover from disturbance and change and in underpinning ecosystem-based adaptation strategies (Colls et al. 2009). Three other key factors are also important in the context of climate-related coastal ecosystem resilience: ecosystem diversity and connectivity (Grimsditch and Salm 2006):

- Functional diversity is particularly important for resilience, for example where grazing herbivores such as reef fish and sea urchins keep macroalgal growth in check, increasing resilience, and increasing recovery from algal phase shifts.
 For example there is increasing evidence that reefs in marine protected areas recover more quickly from disturbances such as cyclones than in unprotected areas.
- For coral reefs there is less evidence that species diversity is as important as functional diversity, although it may provide some redundancy against the risks associated with particular disturbances (e.g., occurrence of more thermally-tolerant coral species). However for mangroves, diversity in species assemblage and zonation over a range of levels is an important resilience feature (McLeod and Salm 2006). Diverse seagrass assemblages are also linked with a higher capacity to adapt to climate change due to the broader range of functional roles and increased tolerance to environmental changes offered by multiple species (Duffy 2006).
- For key coastal tropical habitats, corals, seagrasses and mangroves, connectivity within and between respective ecosystems is important for reproduction and the resilience of habitat diversity. Identifying critical ecosystem connectivity and incorporating these concepts into management approaches, such as marine protected area networks, as a mechanism to increase resilience is a significant challenge that we need to address.

Functional connectivity between ecosystems is also important, with fringing and barrier coral reefs often providing the protective wave-energy environment enabling seagrass and mangrove habitats to develop. In turn mangrove and seagrass help stabilise and bind reef top sediments and buffer coral reefs from land-based sediment and nutrient run-off and resulting sedimentation and water clarity and quality issues. Furthermore seagrass beds in particular may play an extremely important role in locally raising pH levels which may provide a critical local buffer to the effects of ocean acidification to calcifying organisisms that occur within the seagrass beds or closeby.

Multi-functional connectivity is also critical in the coastal protection services provided by these ecosystems in that where occurring together, they: 1) collectively work to progressively reduce wave energy reaching a shoreline reducing the potential for shoreline change, and 2) play key roles in regulating reef flat sediment budgets. Despite the importance of the protective role that these ecosystems play, particularly during normal sea conditions and cyclone events (their protective capacities from tsunami are more variable) information on

this role tends to be observational and anecdotal (UNEP-WCNC 2006). Their effectiveness is difficult to quantify as it is highly site specific and related to a wide range of dynamic and inter-related factors.

Throughout the Pacific region there are an increasing number of ecosystem-based adaptation initiatives, often with a key objective of increasing protection to at-risk coastlines, typically involving replanting and restoring mangrove strands (Figure 6-1). Whilst many of these initiatives will assist in increasing protection levels (as well as provide other ecosystem services), particularly where there is active long-term involvement of local communities in continuing to manage and develop such sites, many of these activities are conducted in relative isolation rather than as one component within the context of:

- addressing the root causes of anthropogenic impacts on the wider coastal ecosystem, which is a more complex and longer-term management issue to tackle. Short-term and small-scale ecosystem rehabilitation activities should not take precedence over addressing these more pervasive background issues (UNEP-WCNC 2006).
- considering, at least qualitatively, the entire natural coastal protection system and the connectivity between and relative influences of the different components.

Fundamentally this includes more than just the protective functions of the coral reef and reef flat associated ecosystems but also needs to include shoreline buffer zones on land. Irrespective of how well such coastal ecosystems provide a protective function, shorelines may still be episodically impacted and if development is built too close to the shoreline or on low-lying land it may still be exposed to coastal hazards. Such land buffers may also enable space for mangrove stands, for example, to migrate landward with sea-level rise.

With any ecosystem-based adaptation, there are barriers and limitations (Colls et al. 2009). In the context of approaches that underpin coastal hazard management, key challenges include:

- Understanding its potential limitations. Such approaches cannot protect coastal communities from all extreme weather events and as climate change increasingly impacts on coastal ecosystem health in turn this will most likely reduce resilience and the level of protective functions provided by such ecosystems.
- Understanding how coastal planning, management and engineering activities and interventions can better add to and complement the protection provided by such ecosystems, rather than detract from it, whilst also enabling adaptive management.



Figure 6-1: Mangroves planted to protect the shoreline along the Ananau Causeway on South Tarawa, Kiribati.

7 Conclusions

Coral reef systems, seagrasses and mangroves in the tropical Pacific and East Timor region provide a fundamental ecosystem service by moderating the inherent dynamic nature of reeffringed shorelines, and the occurrence or severity of episodic storm or swell-related inundation. Coral reefs and associated carbonate producers such as coralline algae and foraminera also provide the primary sources of carbonate framework and sediments for ongoing reef accretion, and the sediments that form many of the reef islands and shorelines across region.

These coastal ecosystems are expected to be particularly susceptible to the additional cumulative stressors of climate change. The climate changes that are currently anticipated to be experienced this century include significant increases in air and ocean temperatures, changes to mean and extreme rainfall and rainfall seasonality, increase in mean sea levels, increase in acidity of ocean waters, and possible changes to winds and currents. Key issues include:

- Coral reef systems are likely to be most sensitive to increased sea surface temperatures and ocean acidification changes, with interactions between temperature and acidification likely to exacerbate impacts for some species. Significant seasurface temperature-related mass coral bleaching events have occurred in the region over recent decades. Many coral species across the region are already near their thermal tolerance, and bleaching events are predicted to be more frequent in coming decades. Similarly ocean acidity is likely to fall below the level required to sustain coral reef accretion within the first half of this century, assuming carbon dioxide emission rates continue at current rates. Both temperature and acidification impacts are likely to occur earlier in the more coral-diverse western Pacific (Melanesia) than in the central mid-ocean areas. Impacts will include:
 - Significant loss of coral species diversity with a favouring of more temperature resilient species.
 - Greater expenditure of energy used for coral skeletal growth resulting in changes to coral metabolism, reproduction and larval settlement.
 - Reduction in calcification rate changing the balance between reef-system accretion and erosion. Reduced rate of coral colony growth and reduced skeletal densities, making coral colonies more prone to damage and reduced resilience to recover from disturbance events. Ultimately long-term reductions in ocean pH will lead to internal and external dissolution of calcified reef structures.
 - Changes in the composition of reef flat carbonate sediments to more thermally tolerant producers, such as *Halimeda* spp. and foraminifera.
- Seagrass and mangrove systems are predicted to be most sensitive to increasing sea-level. For both seagrasses and mangroves, their ability to respond to sea-level rise is determined by the ability of species to colonise and extend shoreward, the availability of suitable substrate, and whether sediment accretion balances erosional

processes. Seagrass and mangroves on islands with fringing reef environments are expected to experience the largest impacts of climate change, especially those with low tide ranges, as higher sea levels enable increased wave energy to propagate over reef flats. Seagrass and mangrove systems in some areas may also experience climate change impacts due to increases in rainfall and storm events, High-island environments are predicted to experience impacts from increased land-based sedimentation due to increased rainfall intensity.

At present many coastal ecosystems, particular those close to population centres, are being significantly impacted by increased sediment and nutrient run-off due to catchment and coastal development, lack of sewage treatment, overfishing, and direct impacts from activities such as dredging and marine pollution. Climate change will exacerbate impacts on coastal ecosystems that are already subject to high levels of human impact. Such impacts significantly decrease the resilience of reef systems and their ability to absorb or recover from a disturbance event.

Climate change and on-going human-related impacts on tropical coastal ecosystems in the Pacific and East Timor will impact on two primary mechanisms of coastal protection that regulate physical change in adjacent shoreline position, and are currently provided by these systems:

- Reducing the amount of wave energy dissipation that occurs due to waves breaking on the outer reef crest and further dissipation of wave energy as waves travel over the reef flat and other coastal habitats.
- Changes in the rate and composition of sediment supply, transfers and storage, and losses of sediments within reef flats and adjacent shorelines.

In many coastal margins, natural features or seawalls and urban development prevent shoreward migration of key coastal ecosystems, such that sea-level rise will result in either increasingly narrow distributions (coastal squeeze) or loss of species zonation. Sea-level rise is expected to be particularly problematic where development or infrastructure located in the coastal margins has reduced the area available for seagrass or mangrove retreat. Under sea-level rise rates expected later this century, mangrove stands on atolls and low islands will be most at risk given the low rates of sedimentation and peat accumulation, lack of sediment supply, and typical lack of available space to move landward.

The coastal protection ability of tropical coastal ecosystems will decrease as climate change impacts reduce the distribution and abundance of key coastal ecosystems. Changes in coastal protection function will be site specific, and depend on complex interactions and feedbacks between these physical processes of wave dissipation and sediment supply, coastal morphology, and current and future impacts of humans on these coastal ecosystems.

Irrespective of the future impacts that climate change will have on coastal ecosystems, these ecosystems currently provide coastal-hazard resilience to adjacent human communities. Improving ecosystem resilience in the context of the entire natural coastal-protection system, and maintaining the flexibility to adapt to change are rarely considered, particularly in the context of addressing coastal hazard issues. However, if adaptation for both Pacific societies and the environment are to be effective, recognising these ecosystems as a fundamental

component of societal health, and addressing the root causes of many of the human-related impacts on them are critical. Even with optimal natural coastal protection functions provided by tropical coastal habitats, there are limitations to the protection service. They cannot protect coastal communities from all extreme weather and storm tide events.

As climate change increasingly impacts on coastal ecosystem health, in turn this will most likely gradually reduce resilience and the level of protective functions provided by such ecosystems. Understanding how coastal planning, management and engineering activities and interventions can better add to and complement the protection provided by such ecosystems, rather than detract from it, is fundamental to increasing the resilience of adjacent communities in the long term.

8 Acknowledgements

We thank Kathryn Julian for performing an extensive literature search for this project. We thank Di Tracey and Helen Bostock for guiding our review of ocean acidification impacts.

9 References

- Adjeroud, M.; Michonneau, F.; Edmunds, P.J.; Chancerelle, Y.; de Loma, T.L.; Penin, L.; Thibaut, L.; Vidal-Dupiol, J.; Salvat, B.; Galzin, R. (2009). Recurrent disturbances, recovery trajectories, and resilience of coral assemblages on a South Central Pacific reef. Coral Reefs 28(3): 775-780.
- Alongi, D.M. (2008). Mangrove forests: Resilience, protection from tsunamis, and responses to global climate change. Estuarine, Coastal and Shelf Science 76: 1-13.
- Baker, A.C. (2004). Diversity, distribution and ecology of Symbiodinium on coral reefs and its relationship to bleaching resistance and resilience. In: Coral health and disease. Edited by E. Rosenberg and Y. Loya. Springer-Verlag, Heidelberg, Berlin. Pp. 177–194.
- Baker, A.C.; Glynn, P.W.; Riegl, B. (2008). Climate change and coral reef bleaching: An ecological assessment of long-term impacts, recovery trends and future outlook. Estuarine Coastal and Shelf Science 80(4): 435-471.
- Ball, M.; Cochrane, M.; Rawson, H. (1997). Growth and water use of the mangroves Rhizophora apiculata and R. stylosa in response to salinity and humidity under ambient and elevated concentrations of atmospheric CO sub(2). Plant, Cell & Environment 20: 1158-1166.
- Beer, S.; Mtolera, M.; Lyimo, T.; Bjork, M. (2006). The photosynthetic performance of the tropical seagrass Halophila ovalis in the upper intertidal. Aquatic Botany 84: 367-371.
- Berkelmans, R. (2002). Time-integrated thermal bleaching thresholds of reefs and their variation on the Great Barrier Reef. Marine Ecology Progress series 237: 309-310.
- Bjork, M.; Short, F.; McLeod, E.; Beer, S. (2008). Managing seagrasses for resilience to climate change, IUCN, Gland, Switzerland.
- Bridges, K.; McMillan, C. (1986). The distribution of seagrasses of Yap, Micronesia, in relation to low tide conditions. Aquatic Botany 24: 403-407.
- Brown, B.E. (1997). Coral bleaching, causes and consequences. Coral Reefs 16 (Suppl.): S129-S138.
- Bruno, J.F.; Selig, E.R. (2007). Regional Decline of Coral Cover in the Indo-Pacific: Timing, Extent, and Subregional Comparisons. Plos One 2(8).
- Buddemeier, R.W.; Fautin, D.G. (1993). Coral bleaching as an adaptive mechanism - A testable hypothesis. BioScience 43: 320-326.
- Burke, L.R.K.; Spalding, M.; Perry, A. (2011). Reefs at Risk Revisited, World Resources Institute, Washington D.C.

- Cahoon, D.R.; Hensel, P.F.; Spencer, T.; Reed, D.J.; McKee, K.; Saintilan, N. (2006). Coastal wetland vulnerability to relative sea-level rise: Wetland elevation trends and process controls. *In* Wetland and Natural Resource Management. . *Edited by* J.T.A. Verhoeven, B. Beltman, R. Bobbink and D.F. Whigham. Springer-Verlag, Berlin. Pp. 271–292.
- Campbell, S.J.; McKenzie, L.J.; Kerville, S.P. (2006). Photosynthetic responses of seven tropical seagrasses to elevated seawater temperature. *Journal of Experimental Marine Biology and Ecology 330*: 455–468.
- Cao, L.; Caldeira, K. (2008). Atmospheric CO₂ stabilization and ocean acidification. *Geophysical Research Letters* 35: L19609.
- Carpenter, K.E.; Abrar, M.; Aeby, G.; Aronson, R.B.; Banks, S.; Bruckner, A.;
 Chiriboga, A.; Cortes, J.; Delbeek, J.C.; DeVantier, L.; Edgar, G.J.; Edwards,
 A.J.; Fenner, D.; Guzman, H.M.; Hoeksema, B.W.; Hodgson, G.; Johan, O.;
 Licuanan, W.Y.; Livingstone, S.R.; Lovell, E.R.; Moore, J.A.; Obura, D.O.;
 Ochavillo, D.; Polidoro, B.A.; Precht, W.F.; Quibilan, M.C.; Reboton, C.; Richards,
 Z.T.; Rogers, A.D.; Sanciangco, J.; Sheppard, A.; Sheppard, C.; Smith, J.; Stuart,
 S.; Turak, E.; Veron, J.E.N.; Wallace, C.; Weil, E.; Wood, E. (2008). One-third of
 reef-building corals face elevated extinction risk from climate change and local
 impacts. . Science 321: 560–563.
- Chen, S.-N.; Sanford, L.P.; Koch, E.W.; Shi, F.; North, E.W. (2007). A nearshore model to investigate the effects of seagrass bed geometry on wave attenuation and suspended sediment transport. *Estuaries 30*: 296–310.
- Clark, T.; Morton, B. (1999). Relative roles of bioerosion and typhoon-induced disturbance on the dynamics of a high latitude scleractinian coral community. *J. Mar. Biol. Assoc. U.K.* 79: 803–820.
- Coles, R.; McKenzie, L., Campbell, S., Fortes, M., and Short, F. 2003. The seagrasses of the western Pacific islands. *In* World atlas of seagrasses. *Edited by* E.P. Green and F.T. Short. University of California Press, Berkeley. Pp. 161– 170.
- Coles, R.G. (1996). Coastal management and Community Coastal Resource Planning in the Asia Pacific Region. .
- Collen, J.D.; Garton, D.W. (2004). Larger foraminifera and sedimentation around Fongafale Island, Funafuti Atoll, Tuvalu Coral reefs 23(3): 445–454.
- Colls, A.; Ash, N.; Ikkala, N. (2009). Ecosystem-based adaptation: a natural response to climate change, Gland, Switzerland.
- Connell, J.H.; Hughes, T.P.; Wallace, C.C. (1997). A 30-year study of coral abundance, recruitment, and disturbance at several scales in space and time. *Ecological Monographs* 67: 461–488.

- Cooper, T.F.; De 'Ath, G.; Fabricius, K.E.; Lough, J.M. (2008). Declining coral calcification in massive Porites in two nearshore regions of the northern Great Barrier Reef. Global Change Biology 14(3): 529-538.
- Costanza, R.: d'Arge, R.: de Groot, R.: Farber, S.: Grasso, M.: Hannon, B.: Limburg, K.; Naeem, S.; O'Neill, R.V.; Paruelo, J.; Raskin, R.G.; Sutton, P.; van den Belt, M. (1997). The value of the world's ecosystem services and natural capital. Nature Australia 387: 253-260.
- Dawson, S.; Dennison, W. (1996). Effects of ultraviolet and photosynthetically active radiation on five seagrass species. Marine Biology 125(4): 629-638.
- Department of Climate Change and Energy Efficiency (2011). The critical decade: Climate science, risks and responses.
- Diaz-Pulido, G.; McCook, L.J.; Dove, S.; Berkelmans, R.; Roff, G.; Kline, D.I.; Weeks, S.; Evans, R.D.; Williamson, D.H.; Hoegh-Guldberg, O. (2009). Doom and Boom on a Resilient Reef: Climate Change, Algal Overgrowth and Coral Recovery. Plos One 4(4).
- Diaz-Pulido, G.; McCook, L.J.; Larkum, A.W.D.; Lotze, H.K.; Raven, J.A.; Schaffelke, B.; Smith, J.E.; Steneck, R.S. (2007). Vulnerability of macroalgae of the Great Barrier Reef to climate change. In: Climate Change and the Great Barrier Reef, . Edited by J.E. Johnson and P.A. Marshall. Great Barrier Reef Marine Park Authority and Australian Greenhouse Office, Townsville, Australia. Pp. 153–192.
- Done, T.; Jones, R. (2006). Tropical coastal ecosystems and climate change prediction: global and local risks. In: Coral reefs and climate change: science and management. Coastal and Estuarine Studies 61. . Edited by J.T. Phinney, O. Hoegh-Guldberg, J. Kleypas, W. Skirving and A. Strong. American Geophysical Union. pp. 5-32.
- Done, T.J. (1992). Phase shifts in coral reef communities and their ecological significance. Hydrobiologia 247: 121–132.
- Donnelly, J.; Bertness, M. (2001). Rapid shoreward encroachment of salt marsh cordgrass in response to accelerated sea-level rise. PNAS 98(25): 14218-14223.
- Donner, S.D. (2009). Coping with commitment: Projecting future thermal stress on coral reefs worldwide and the potential importance of the Central Equatorial Pacific Proceedings of the 2009 American Geophysical Union Joint Assembly.
- Doody, J.P. (2004). 'Coastal squeeze' an historical perspective. Journal of Coastal Conservation, 10/1-2: 129-138.
- Duarte, C.M.; Chiscano, C.L. (1999). Seagrass biomass and production: A reassessment. Aquatic Botany 65: 159-174.
- Duffy, J.E. (2006). Biodiversity and the functioning of seagrass ecosystems. Marine Ecology - Progress Series 311: 233–250.

- Eakin, C.M.; Morgan, J.A.; Heron, S.F.; Smith, T.B.; Liu, G.; Alvarez-Filip, L.; Baca, B.; Bartels, E.; Bastidas, C.; Bouchon, C.; Brandt, M.; Bruckner, A.W.; Bunkley-Williams, L.; Cameron, A.; Causey, B.D.; Chiappone, M.; Christensen, T.R.L.; Crabbe, M.J.C.; Day, O.; de la Guardia, E.; Diaz-Pulido, G.; DiResta, D.; Gil-Agudelo, D.L.; Gilliam, D.S.; Ginsburg, R.N.; Gore, S.; Guzman, H.M.; Hendee, J.C.; Hernandez-Delgado, E.A.; Husain, E.; Jeffrey, C.F.G.; Jones, R.J.; Jordan-Dahlgren, E.; Kaufman, L.S.; Kline, D.I.; Kramer, P.A.; Lang, J.C.; Lirman, D.; Mallela, J.; Manfrino, C.; Marechal, J.P.; Marks, K.; Mihaly, J.; Miller, W.J.; Mueller, E.M.; Muller, E.M.; Toro, C.A.O.; Oxenford, H.A.; Ponce-Taylor, D.; Quinn, N.; Ritchie, K.B.; Rodriguez, S.; Ramirez, A.R.; Romano, S.; Samhouri, J.F.; Sanchez, J.A.; Schmahl, G.P.; Shank, B.V.; Skirving, W.J.; Steiner, S.C.C.; Villamizar, E.; Walsh, S.M.; Walter, C.; Weil, E.; Williams, E.H.; Roberson, K.W.; Yusuf, Y. (2010). Caribbean Corals in Crisis: Record Thermal Stress, Bleaching, and Mortality in 2005. *Plos One* 5(11).
- Ellison, J. (1993). Mangrove retreat with rising sea-level, Bermuda. *Estuarine, Coastal and Shelf Science* 37(1): 75–87.
- Ellison, J. (1996). Potential impacts of predicted climate change on mangroves: Implications for marine parks. *Parks (International Journal for Protected Area Management (IUCN))* 6(2): 14–24.
- Ellison, J. (1999). Status report on Pacific island mangroves. *In:* Marine and coastal biodiversity in the tropical island Pacific region: impacts and responses in Australia, New Zealand, and small island states. *Edited by* L.G. Eldredge, J.E. Maragos and P.L. Holthus. Kluwer Academic Publishers, Dordrecht, Netherlands. Pp. 3–19.
- Ellison, J. (2004). Vulnerability of Fiji's Mangroves and Associated Coral Reefs to Climate Change, World Wildlife Fund, Launceston, Australia: University of Tasmania.
- Ellison, J.C. (1997). Mangrove ecosystems of the Western and Gulf Provinces, Papua New Guinea, a review. . *Science in New Guinea 23*: 1–15.
- Ellison, J.C. (2009). Wetlands of the Pacific Island region. *Wetlands Ecology and Management 17(3):* 169–206.
- Ewel, K.C.; Bourgeois, J.; Cole, T.; Zheng, S. (1998). Variation in environmental characteristics and vegetation in high-rainfall mangrove forests, Kosrae, Micronesia. *Global Ecology and Biogeography Letters* 7: 49–56.
- Fabricius, K.E.; De'ath, G.; Puotinen, M.L.; Done, T.; Cooper, T.F.; Burgess, S.C. (2008). Disturbance gradients on inshore and offshore coral reefs caused by a severe tropical cyclone. *Limnology and Oceanography* 53(2): 690–704.
- Fabricius, K.E.; Langdon, C.; Uthicke, S.; Humphrey, C.; Noonan, S.; De'Ath, G.; Okazaki, R.; Muehllehner, N.; Glas, M.S.; Lough, J.M. (2011). Losers and winners in coral reefs acclimatized to elevated carbon dioxide concentrations. *Nature Climate Change 1*: 165–169.

- Farnsworth, E.; Ellison, A.; Gong, W. (1996). Elevated CO₂ alters anatomy, physiology, growth, and reproduction of red mangrove (*Rhizophora mangle* L.) *Oecologia* 108(4): 599–609.
- Fitt, W.K.; McFarland, F.K.; Warner, M.E.; Chilcoat, G.C. (2000). Seasonal patterns of tissue biomass and densities of symbiotic dinoflagellates in reef corals and relation to coral bleaching. *Limnology and Oceanography* 45(3): 677–685.
- Fong, P.; Harwell, M. (1994). Modelling seagrass communities in tropical and subtropical bays and estuaries: A mathematical synthesis of current hypotheses. *Bulletin of Marine Science* 54: 757–781.
- Fonseca, M.S.; Cahalan, J.A. (1992). A preliminary evaluation of wave attenuation for four species of seagrass. *Estuarine, Coastal and Shelf Science* 35: 565–576.
- Fuentes, M.M.P.B.; Dawson, J.; Smithers, S.; Limpus, C.J.; Hamann, M. (2010). Sedimentological characteristics of key sea turtle rookeries: potential implications under projected climate change. Journal of Marine and Freshwater Research 61: 464–473.
- Fujita, K.; Hikami, M.; Suzuki, A.; Kuroyanagi, A.; Kawahata, H. (2011). Effects of ocean acidification on calcification of symbiont-bearing reef foraminifers. *Biogeosciences Discussions 8*: 1809–1829.
- Gilman, E.L. (1999). Chapter 1. Compensatory wetland mitigation in the CNMI: An incipient discipline. *In:* UNESCO World Heritage Central Pacific Project Planning Workshop, 5-9 October 2004, Kiritimati Island, Republic of Kiribati. Pp. 3–17.
- Gilman, E.L.; Ellison, J.; Duke, N.C.; Field, C. (2008). Threats to mangroves from climate change and adaptation options: A review. *Aquatic Botany 89(2)*: 237–250.
- Gilman, E.L.; Ellison, J.; Jungblut, V.; Van Lavieren, H.; Wilson, L.; Areki, F.;
 Brighouse, G.; Bungitak, J.; Dus, E.; Henry, M.; Kilman, M.; Matthews, E.; Sauni, L.; Teariki-Ruatu, N.; Tukia, S.; Yuknavage, K. (2006). Adapting to Pacific Island mangrove responses to sea level rise and climate change. *Climate Research* 32(3): 161–176.
- Goldberg, J.A., K.; Albert, J.; Asher, J.; Brown, P.; Brown, V.; Burdick, D.; Carroll, B.; Craig, P.; Fenner, D., Fillmed, C.; Fread, V.; Gawel, M.; George, A.; Golbuu, Y.; Goldman, L.; Graham, C.; Hall, A.; Hasurmai, M.; Jacob, L.; Jacobson, D.; Joseph, E.; Kenyon, J.; Kostka, W.; Leberer, T.; Luckymis, M.; Lundblad, E.; Malakai, S.; Maragos, J.; Marcus, A.; Marino, S.; Mathias, D.; McIlwain, J.; Miller, J.; Minton, D.; Nadon, M.; Palik, S.; Pioppi, N.; Raymundo, L.; Richards, B.; Sabater, M.; Schroeder, R.; Schupp, P.; Smith, E.; Tyakesy, A.; Zgliczynski, B. (2008). Status of Coral Reef Resources in Micronesia and American Samoa: 2008, Global Coral Reef Monitoring Network, Townsville.

Graham, N.A.J.; Wilson, S.K.; Jennings, S.; Polunin, N.V.C.; Robinson, J.; Bijoux, J.P.; Daw, T.M. (2007). Lag effects in the impacts of mass coral bleaching on

coral reef fish, fisheries, and ecosystems. *Conservation Biology 21(5)*: 1291–1300.

- Green, E.P.; Short, F.T. (2003). World atlas of seagrasses. University of California Press, Berkeley.
- Grimsditch, G.D.; Salm, R.V. (2006). Coral reef resilience and resistance to bleaching, IUCN, Gland, Switzerland.
- Guinotte, J.M.; Buddemeier, R.W.; Kleypas, J.A. (2003). Future coral reef habitat marginality: temporal and spatial effects of climate change in the Pacific basin. *Coral Reefs* 22(4): 551–558.
- Guinotte, J.M.; Fabry, V.J. (2008). Ocean acidification and its potential effects on marine ecosystems. *Year in Ecology and Conservation Biology 2008 1134*: 320–342.
- Guinotte, J.M.; Orr, J.; Cairns, S.; Freiwald, A.; Morgan, L.; George, R. (2006). Will human induced changes in seawater chemistry alter the distribution of deep-sea scleractinian corals? . *Frontiers in Ecology and Environment 4*: 141–146.
- Hart, D.E.; Kench, P.S. (2007). Carbonate production of an emergent reef platform, Warraber Island, Torres Strait, Australia. *Coral Reefs 26*: 53–68.
- He, J.-S.; Bazzaz, F. (2003). Density-dependent responses of reproductive allocation to elevated atmospheric CO₂ in *Phytolacca americana*. *New Phytologist 157(2)*: 229–239.
- Hemminga, M.; Duarte, C.M. (2000). Seagrass Ecology. Cambridge University Press, Cambridge (United Kingdom).
- Hoegh-Guldberg, O. (1999). Climate change, coral bleaching and the future of the world's coral reefs. *Marine and Freshwater Research 50(8)*: 839–866.
- Hoegh-Guldberg, O. (2009a). Climate change and coral reefs: Trojan horse or false prophecy? *Coral Reefs* 28(3): 569–575.
- Hoegh-Guldberg, O.; Hughes, L.; McIntyre, S.; Lindenmayer, D.B.; Parmesan, C.; Possingham, H.P.; Thomas, C.D. (2008). Assisted colonization and rapid climate change. *Science* 321: 345–346.
- Hoegh-Guldberg, O.; Mumby, P.J.; Hooten, A.J.; Steneck, R.S.; Greenfield, P.;
 Gomez, E.; Harvell, C.D.; Sale, P.F.; Edwards, A.J.; Caldeira, K.; Knowlton, N.;
 Eakin, C.M.; Iglesias-Prieto, R.; Muthiga, N.; Bradbury, R.H.; Dubi, A.; Hatziolos,
 M.E. (2007). Coral reefs under rapid climate change and ocean acidification. *Science 318(5857)*: 1737–1742.
- Hoegh-Guldberg, O.; Salvat, B. (1995). Periodic mass-bleaching and elevated sea temperatures: bleaching of outer reef slope communities in Moorea, French Polynesia. *Marine Ecology Progress Series 121*: 181–190.

- Hoegh-Guldberg, O.L.Y.; Bythell, J.; Fitt, W.; Gates, R.; Iglesiasa-Prieto, R.; Lesser, M.; McClanahan, T.; van Woesik, R.; Wild, C. (2009b). Bleaching and Related Ecological Factors: CRTR Working Group Findings 2004-2009, Coral Reef Targeted Research & Capacity Building for Management.
- Houk, P.; Camacho, R. (2010). Dynamics of seagrass and macroalgal assemblages in Saipan Lagoon, Western Pacific Ocean: disturbances, pollution, and seasonal cycles. Botanica Marina 53(3): 205-212.
- Hughes, T.; Connell, J. (1999). Multiple stressors on coral reefs: A long-term perspective. Limnology and Oceanography 44(3): 932-940.
- Hughes, T.; Szmant, A.M.; Steneck, R.; Carpenter, R.; Miller, S. (1999). Algal blooms on coral reefs: what are the causes? Limnology and Oceanography 44(6): 1583-1586.
- Hughes, T.P.; Baird, A.H.; Bellwood, D.R.; Card, M.; Connolly, S.R.; Folke, C.; Grosberg, R.; Hoegh-Guldberg, O.; Jackson, J.B.C.; Kleypas, J.; Lough, J.M.; Marshall, P.; Nystrom, M.; Palumbi, S.R.; Pandolfi, J.M.; Rosen, B.; Roughgarden, J. (2003). Climate change, human impacts, and the resilience of coral reefs. Science 301(5635): 929-933.
- Hughes, T.P.; Graham, N.A.J.; Jackson, J.B.C.; Mumby, P.J.; Steneck, R.S. (2010). Rising to the challenge of sustaining coral reef resilience. Trends in Ecology & Evolution 25(11): 633-642.
- IPCC (2007). Climate Change 2007: Impacts, Adaptation and Vulnerability.
- Jaffar, M. (1992). Country report on mangrove ecosystems in the Republic of Fiji. Proceedings of the Seminar and Workshop on integrated research on mangrove ecosystems in the Pacific Islands region II.
- Jokiel, P.L.; Rodgers, K.S.; Kuffner, I.B.; Andersson, A.J.; Cox, E.F.; Mackenzie, F.T. (2008). Ocean acidification and calcifying reef organisms: a mesocosm investigation. Coral Reefs 27: 473-483.
- Kench, P.; Perry, C.; Spencer, T. (2009). Coral reefs. In: Geomorphology and Global Environmental Change. Edited by O. Slaymaker, T. Spencer and C. Embleton-Hamann. Cambridge University Press, Cambridge. Pp. 180–213.
- Kjerfve, B. (1990). Manual for investigation of hydrological processes in mangrove ecosystems, UNESCO/UNDP.
- Kleypas, J.A.; Yates, K.K. (2009). Coral reefs and ocean acidification. Oceanography 22(4): 108–117.
- Koch, E.W.; Ackerman, J.; van Keulen, M. (2006). Fluid dynamics in seagrass ecology: from molecules to ecosystems. In: Seagrasses: biology, ecology and their conservation. Edited by L.A., O.R. and D.C. Springer-Verlag, Berlin, Germany.

- Krauss, K.W.; Allen, J.A.; Cahoon, D.R. (2003). Differential rates of vertical accretion and elevation changes among aerial root types in Micronesian mangrove forests. *Estuarine Coastal and Shelf Science 5*4: 251–259.
- Kuffner, I.B.; Andersson, A.J.; Jokiel, P.L.; Rodgers, K.U.S.; MacKenzie, F.T. (2008). Decreased abundance of crustose coralline algae due to ocean acidification. *Nature Geoscience* 1(2): 114–117.
- Lacambra, C.S.T.; Moeller, I. (2008). Literature Review: Tropical Coastal Ecosystems As Coastal Defences. *In:* The Role of Environmental Management and Eco-Engineering in Disaster Risk Reduction and Climate Change Adaptation, Cambridge Coastal Research Unit, Department of Geography, University of Cambridge, United Kingdom. Pp. 1–22.
- Langdon, C. (2002). Review of experimental evidence for effects of CO2 on calcification of reef-builders Proceedings 9th International Coral Reef Symposium, Bali, Indonesia. Pp. 1091–1098.
- Littler, M.M.; Littler, D.S. (1994). Tropical reefs as complex habitats for diverse macroalgae. *In:* Seaweed ecology and physiology. *Edited by* C.S.H. Lobban, P.J. Cambridge University Press, New York. Pp. 72–75.
- Lovelock, C.E.; Ellison, J. (2007). Vulnerability of mangroves and tidal wetlands of the Great Barrier Reef to climate change. *In:* Climate Change and the Great Barrier Reef . *Edited by* J.E. Johnson and P.A. Marshall. Great Barrier Reef Marine Park Authority and Australian Greenhouse Office, Townsville, Australia. Pp. 237–270.
- Maragos, J.E. (1993). Impact of coastal construction on coral reefs in the US affiliated Pacific islands. *Coastal Management 21*: 235–269.
- Massel, S.R.; Furukawa, K.; Brinkman, R.M. (1999). Surface wave propagation in mangrove forests. *Fluid Dynamics Research 24*: 219–249.
- McClanahan, T.R.; Baird, A.H.; Marshall, P.A. (2004). Comparing bleaching and mortality responses of hard corals between southern Kenya and the Great Barrier Reef, Australia. *Marine Pollution Bulletin 48*: 327–335.
- McClanahan, T.R.; Muthiga, N.A. (1988). Changes in Kenyan coral reef community structure and function due to exploitation. *Hydrobiologia 166*: 269–276.
- McCook, L.J. (1999). Macroalgae, nutrients, and phase shifts on coral reefs: scientific issues and management consequences for the Great Barrier Reef. *Coral Reefs 18*: 357–367.
- McKenzie, L.J.; Mellors, J.E.; Yoshida, R.L.; Coles, R.G. 2006. Seagrass-Watch: A non-destructive, seagrass assessment and monitoring program 1998-2006.
 Abstracts: Catchments to Coast. Cairns Qld Australia 9-14 July 2006. Australian Marine Sciences Association 44th Annual Conference and the Society of Wetland Scientists 27th International Conference. P. 73.

- McKenzie, L.J.; Yoshida, R.L. (2007). Seagrass-Watch: Guidelines for Monitoring Seagrass Habitats in the Fiji Islands. Proceedings of a training workshop, Corpus Christi Teachers College, Laucala Bay, Suva, Fiji, 16th June 2007, Seagrass-Watch HQ, Cairns, Australia.
- McLeod, E.; Salm, R.V. (2006). Managing mangroves for resilience to climate change, IUCN, Gland, Switzerland.
- McMillian, C. (1984). The distribution of tropical seagrasses with relation to their tolerance of high temperatures. Aquatic Botany 19: 369-379.
- Mumby, P.J.; Dahlgren, C.P.; Harborne, A.R.; Kappel, C.V.; Micheli, F.; Brumbaugh, D.R.; Holmes, K.E.; Mendes, J.M.; Broad, K.; Sanchirico, J.N.; Buch, K.; Box, S.; Stoffle, R.W.; Gill, A.B. (2006). Fishing, trophic cascades, and the process of grazing on coral reefs. Science 311: 98-101.
- Nelson, W.A. (2009). Calcified macroalgae critical to coastal ecosystems and vulnerable to change: a review. Marine and Freshwater Research 60(8): 787-801.
- Neumann, A.C.; Macintyre, I. (1985). Reef response to sea level rise: keep-up, catch-up or give-up. . Proceedings of the Fifth International Coral Reef Symposium: 105–110.
- Obura, D. (2005). Resilience and climate change: Lessons from coral reefs and bleaching in the Western Indian Ocean. Estuarine and Coastal Marine Science 63 (353-372).
- Oliver, J.K.; Berkelmans, R.; Eakin, C.M. (2009). Coral Bleaching in Space and Time. Ecological Studies 205: 21-39.
- Orth, R.J.; Carruthers, T.J.B.; Dennison, W.C.; Duarte, C.M.; Fourgurean, J.W.; Heck, K.L. Jr.; Hughes, A.R.; Kendrick, G.A.; Kenworthy, W.J.; Olyarnik, S.; Short, F.T.; Waycott, M.; Williams, S.L. (2006). A global crisis for seagrass ecosystems. BioScience 56(12): 987-996.
- ProAct-Network (2008). The role of environmental management and ecoengineering in disaster risk reduction and climate change adaptation.
- Quartel, S.; Kroon, A.; Augustinus, P.G.E.F.; Van Santen, P.; Tri, N.H. (2007). Wave attenuation in coastal mangroves in the Red River Delta, Vietnam. Journal of Asian Earth Sciences 29: 576–584.
- Ralph, P.J. (1997). Photoinhibitory stress physiology of the seagrass, Halophila ovalis (R. Br.) Hook. Department of Environmental Biology and Horticulture, University of Technology, Sydney, Australia.
- Saintilan, N.; Wilton, K. (2001). Changes in the distribution of mangroves and saltmarshes in Jervis Bay, Australia. Wetlands Ecology and Management 9(5): 409-420.

- Sathirathai, S.; Barbier, E.B. (2001). Valuing mangrove conservation in southern Thailand. *Contemporary Economic Policy 19(2)*: 109–122.
- Scott, A.X.; Rotondo, G.M. (1983). A model to explain the differences between Pacific plate island-atoll types. *Coral Reefs 1*: 139–150.
- Sebens, K.P. (1994). Biodiversity of coral reefs: What are we losing and why? *American Zoologist 34(1)*: 115–133.
- Semesi, I.S.; Beer, S.; Bjork, M. (2009). Seagrass photosynthesis controls rates of calcification and photosynthesis of calcareous macroalgae in a tropical seagrass meadow. *Marine Ecology Progress Series* 382: 41–47.
- Sheppard, C.; Dixon, D.J.; Gourlay, M.; Sheppard, A.; Payet, R. (2005). Coral mortality increases wave energy reaching shores protected by reef flats: Examples from the Seychelles. Estuarine, *Coastal and Shelf Science 64(2-3)*: 223–234.
- Short, F.; Burdick, D.; Short, C.; Davies, R.C.; Morgan, P.A. (2000). Developing success criteria for restored eelgrass, salt marsh and mud flat habitats. *Ecological Engineering* 15: 239–252.
- Short, F.T.; Neckles, H.A. (1999). The effects of global climate change on seagrasses. *Aquatic Botany 63(3-4)*: 169–196.
- Silverman, J.; Lazar, B.; Cao, L.; Caldeira, K.; Erez, J. (2009). Coral reefs may start dissolving when atmospheric CO₂ doubles. *Geophysical Research Letters* 36: L05606.
- Solomon, S.M.; Forbes, D.L. (1999). Coastal hazards and associated management issues on South Pacific Islands. *Ocean & Coastal Management 42(6-7)*: 523–554.
- South, R.; Skelton, P. (2000). Status of coral reefs in the southwest Pacific: Fiji, Nauru, New Caledonia, Samoa, Solomon Islands, Tuvalu and Vanuatu. *In:* Status of Coral Reefs of the World, 2000. Pp. 159–180.
- Spalding, M.; Kainuma, M.; Collins, L. (2010). World atlas of mangroves. Earthscan, London.
- Spalding, M.; Taylor, M.; Ravilious, C.; Short, F.; Green, E. (2003). Global overview: The distribution and status of seagrasses. *In* World atlas of seagrasses. *Edited by* E.P. Green and F.T. Short. University of California Press, Berkeley. Pp. 5–26.
- Spalding, M.D.; Ravilious, C.; Green, E.P. (2001). World atlas of coral reefs. University of Cailfornia Press, Berkeley.
- Spencer, T. (1995). Potentialities, uncertainties and complexities in the response of coral reefs to future sea-level rise. *Earth Surface Processes and Landforms* 20(1): 49–64.

- Stone, L.; Huppert, A.; Rajagopalan, B.; Bhasin, H.; Loya, Y. (1999). Mass coral reef bleaching: a recent outcome of increased El Niño activity? Ecology Letters 2: 325-330.
- Sullivan, M. (1990). The impacts of projected climate change on coastal land use in Papua New Guinea. In: Implications of expected climatic changes in the South Pacific region: an overview. Edited by J.C. Pernetta and P.J. Hughes. UNEP (UNEP Regional Sea Reports and Studies 128). Nairobi. Pp. 33-58.
- Szmant, A.M.; Gassman, N.J. (1990). The effect of prolonged "bleaching" on the tissue biomass and reproduction of the reef coral Montastrea annularis. Coral Reefs 8: 217-224.
- Ulstrup, K.E.; Berkelmans, R.; Ralph, P.; Van Oppen, M.J. (2006). Variation in bleaching sensitivity of two coral species across a latitudinal gradient on the Great Barrier Reef: the role of zooxanthellae. Marine Ecology Progress Series 314: 135-148.
- UNEP-WCNC (2006). In the front line: shoreline protection and other ecosystem services from mangroves and coral reefs, UNEP-WCMC, Cambridge, UK.
- Veron, J.E.N. (2008). Mass extinctions and ocean acidification: biological constraints on geological dilemmas. Coral Reefs 27(3): 459-472.
- Ward, L.G.; Kemp, W.M.; Boynton, W.E. (1984). The influence of waves and seagrass communities on suspended particulates in an estuarine embayment. . Marine Geology 59: 85–103.
- Watling, D. (1985). A mangrove management plan for Fiji (Phase 1), South Pacific Commission and Fiji government.
- Waycott, M.; Collier, C.; McMahon, K.; Ralph, P.; McKenzie, L.; Udy, J.; Grech, A. (2007). Vulnerability of seagrasses in the Great Barrier Reef to climate change. In Climate Change and the Great Barrier Reef. Edited by J.E. Johnson and P.A. Marshall. Great Barrier Reef Marine Park Authority and Australian Greenhouse Office, Townsville, Australia. Pp. 193-236.
- Whitesell, C.D.; MacLean, C.D.; Falanruw, M.C.; Cole, T.G.; Ambacher, A.H. (1986). Vegetation survey of Kosrae, Federated States of Micronesia. Resource Bulletin PSW 17: 1-18.
- Wilkinson, C.R. (2002). Status of coral reefs of the world, Australian Institute of Marine Science, Townsville, Australia.
- Woodridge, S.; Done, T.J. (2004). Learning to predict large-scale coral bleaching from past events. A Bayesian approach using remotely sensed data, in-situ data, and environmental proxies. Coral Reefs 23: 96-108.
- Woodroffe, C.D. (2002). Reef-island sedimentation on Indo-Pacific atolls and platform reefs. . Proceedings of the 9th International Coral Reef Symposium 2: 1187-1192.
- Yamano, H.; Kayanne, H.; Chikamori, M. (2005). An overview of the nature and dynamics of reef islands. *Global Environmental Research 9(1)*: 9–20.
- Zann, L.P. (1994). The status of coral reefs in South-western Pacific Islands. *Marine Pollution Bulletin 29(1-3)*: 52–61.