Marine and Tropical Sciences Research Facility

Synthesis Report

Water quality and climate change: Managing for resilience




Compiled by J. Johnson and K. Martin

Australian Government
Department of the Environment, Water, Heritage and the Arts

Reef & Rainforest Research Centre

MTSRF Synthesis Report

Water quality and climate change:

Managing for resilience

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Australian Government

Department of Sustainability, Environment, Water, Population and Communities

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The Australian Government's Marine and Tropical Sciences Research Facility (MTSRF) supports world-class, public good research. The MTSRF is a major initiative of the Australian Government, designed to ensure that Australia's environmental challenges are addressed in an innovative, collaborative and sustainable way. The MTSRF investment is managed by the Department of Sustainability, Environment, Water, Population and Communities, and is supplemented by substantial cash and in-kind investments from research providers and interested third parties. The Reef and Rainforest Research Centre Limited (RRRC) is contracted by the Department to provide program management and communications services for the MTSRF.

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Reef & Rainforest
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Thanks are also extended to the then Commonwealth Department of the Environment, Water, Heritage and the Arts (DEWHA) for funding the MTSRF, and the Reef and Rainforest Research Centre (RRRC) for supporting this program over the past four years. The ongoing involvement of primary end users in this research, including DEWHA, the Great Barrier Reef Marine Park Authority (GBRMPA), WWF, the Queensland Department of the Premier and Cabinet (DPC) and regional Natural Resource Management (NRM) groups has focused the research outputs, such that they are already being applied to management activities.

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Acronyms and Abbreviations

COTS .... Crown-of-thorns Starfish

DEWHA............. Commonwealth Department of the Environment, Water, Heritage and the Arts (now Sustainability, Environment, Water, Population and Communities)

DIN .... Dissolved Inorganic Nitrogen

DIP .... Dissolved Inorganic Phosphorus

DON .... Dissolved Organic Nitrogen

DPC................... Queensland Department of the Premier and Cabinet

GBR .... Great Barrier Reef

GBRMPA .... Great Barrier Reef Marine Park Authority

MODIS............... Moderate Resolution Imaging Spectroradiometer

MTSRF .... Marine and Tropical Sciences Research Facility

NRM .... Natural Resource Management

RRRC................ Reef and Rainforest Research Centre

SST .... Sea Surface Temperature

WWF .... World Wide Fund for Nature

About this Report

This report provides a synthesis of the key findings of research conducted under the Australian Government's Marine and Tropical Sciences Research Facility (MTSRF) relevant to understanding interactions between water quality and climate change, and how best to manage for resilience. The report summarises the findings of projects supported by the MTSRF. Some of the information in this report is extracted from these project reports with the permission of the authors.

A key achievement of the MTSRF has been strong cooperation and collaboration between research institutions in project development and implementation. The findings presented in this report were derived from collaborative projects funded from several sources including the MTSRF, and the research institutions have also contributed significant in-kind resources. It should be noted that supporting information external to the MTSRF is included in this report to provide context or to complete the discussion. Publications generated specifically by the MTSRF are identified in the References section.

This synthesis report is one product in a series of informative reports that summarise MTSRF research findings relevant to the GBR. Other products include:

- 'Managing Great Barrier Reef water quality: MTSRF research overview', compiled by J. Waterhouse and M. Devlin (Waterhouse and Devlin 2011).
- 'Improved understanding of biophysical and socio-economic connections between catchment and reef ecosystems: Wet and Dry Tropics case studies', compiled by M. Devlin and J. Waterhouse (Devlin and Waterhouse 2011).
- 'Optimising water quality and impact monitoring, evaluation and reporting programs', compiled by J. Waterhouse (Waterhouse 2010a).
- 'Thresholds of major pollutants with regard to impacts on instream and marine ecosystems', compiled by J. Waterhouse (Waterhouse 2010b)
- 'Identification of priority pollutants and priority areas in the Great Barrier Reef catchments', compiled by J. Waterhouse and J. Brodie (Waterhouse and Brodie 2011).
- 'Managing for GBR resilience: Socio-economic influences', compiled by J. Johnson and K. Martin (Johnson and Martin 2011).
- 'Review of MTSRF research advancing our understanding of the source, transport and impacts of pesticides on the Great Barrier Reef and associated ecological systems', compiled by M. Devlin and S. Lewis (Devlin and Lewis 2011).

Executive Summary

Climate change has been recognised as one of the greatest threats to coral reefs around the world, including the Great Barrier Reef (GBR). A reduction in global carbon emissions is the only long-term solution to the problems caused by climate change. However, effective management and strategic investment in enhancing reef resilience, such as improvements to water quality, will help mitigate some of the risks in the short term, giving natural communities time to adapt or acclimatise to a changing environment.

Research funded by the Australian Government's Marine and Tropical Sciences Research Facility (MTSRF) has increased our understanding of the response of GBR ecosystems to predicted increases in the atmospheric concentration of greenhouse gasses (mainly CO_2), such as increasing sea surface temperature, ocean acidification and secondary influences of increasing incidents of coral disease. In addition, the complex interactions between climate change, ocean acidification and water quality, and the mechanisms behind how water quality influences reef resilience have been examined closely and new knowledge has emerged. Collectively, this new information has been used by MTSRF researchers to develop models that can deal with the complex nature of these relationships to predict how reefs might respond to increasing severity of thermal events, increasing incidents of coral disease, ocean acidification and ultimately the interactions between ecosystem health, water quality and resilience.

While many questions still remain unanswered, findings to date have provided important direction and tools for management to prioritise and target effort, such as knowledge to identify the features of resilient areas of the GBR as possible future refugia that may facilitate recovery of less resilience reefs. This knowledge also provides further impetus to strengthen and implement strategies already aimed at improving GBR water quality, as this remains a significant area where intervention can have tangible effects on resilience to future climate change.

It is also recognised that any decline in the health of inshore GBR habitats will have social and economic implications for coastal communities and reef-dependent industries. There is therefore a need to engage with GBR communities to increase awareness of the issues and continue to address land management practices that improve inshore water quality and enhance resilience of the GBR ecosystem (see the companion report 'Managing for GBR resilience: Socio-economic influences' by Johnson and Martin, 2011).

1. Introduction

The Great Barrier Reef (GBR) is likely to experience increasing frequency and intensity of disturbances under future global change, driven by increasing sea surface temperature (SST), ocean acidification, changing ocean circulation, storm intensity and sea level rise. The GBR Outlook Report (GBRMPA 2009) states that the "... overall outlook for the Great Barrier Reef is poor and catastrophic damage to the ecosystem may not be averted". Climate change was assessed as the dominant factor influencing the future prospects for the GBR.

Fundamental ecological processes on the GBR are already being affected by climate change; higher SSTs have resulted in three major bleaching events in the last decade, and increasing ocean acidity has been identified as affecting calcification rates in reef-building species (GBRMPA 2009). The extent and persistence of damage to the ecosystem will depend to a large degree on the rate and magnitude of future change, and the resilience of the GBR ecosystem (GBRMPA 2009). While influencing the rate of change of the world's climate is beyond the scope of local or regional management, research funded through the Marine and Tropical Sciences Research Facility (MTSRF) has shown that management can influence the resilience of reef ecosystems, with the goal of reducing the likelihood of the system changing to an undesirable state.

Ecological resilience is the "capacity of an ecosystem to resist, recover or regenerate from disturbances or damage without a change in state, so as to maintain key functions and processes" (Nyström *et al.* 2000). The health of the GBR depends on the integrity of its ecological processes, and may be characterised by high diversity and species richness or its resilience to anthropogenic and natural disturbance.

Declining water quality, particularly in the inshore GBR, is one pressure that undermines reef resilience and is likely to be exacerbated by ocean warming and acidification (Anthony *et al.* 2011). Agricultural, urban and industrial development in the GBR catchment has significantly increased the sediment, nutrient and pesticide run-off from rivers draining into coastal and inshore waters (Rayment and Neil 1997, Furnas 2003, McCulloch *et al.* 2003, Fabricius 2005, Brodie *et al.* 2008). Numerous field observations and laboratory studies have shown that nutrient enrichment, turbidity, sedimentation and pesticides all affect the resilience of the GBR ecosystem, degrading coral reefs and seagrass meadows at local and regional scales (e.g. Fabricius 2010, Waycott and McKenzie 2010).

MTSRF-funded research has contributed significantly to the general view that GBR ecosystems decline in species richness and diversity with increasing exposure to nutrients, sediments and pesticides, compromising their ability to maintain essential ecosystem functions with increasing frequency of disturbances. Inshore ecosystems are most at risk from pollutants delivered during small to medium flood events, with risk related to the predominant land-use of the catchment. For example, areas with significant fertiliser use have been identified as regions associated with urban development and high intensity land-uses, such as fertilised crops in coastal areas of the Wet Tropics, Lower Burdekin, Mackay Whitsunday and Burnett Mary catchments (Brodie *et al.* 2008, 2009a, b). Most sediment entering the GBR comes from catchments with large pastoral areas such as the Burdekin and Fitzroy Rivers (Brodie *et al.* 2009b). Pesticides from agricultural activities are present in the GBR lagoon in both wet and dry seasons (see Prange *et al.* 2009, Johnson *et al.* 2010, Johnson *et al.* 2011), and recent monitoring data suggest that pesticides have become ubiquitous contaminants throughout the GBR (Bartkow *et al.* 2008, McKenzie *et al.* 2010, Lewis *et al.* 2009, Bainbridge *et al.* 2009a, Packett *et al.* 2009, Shaw *et al.* 2010).

The targeted, solutions-based research conducted under MTSRF means that the responses of GBR coral reef communities to turbidity and nutrients from terrestrial run-off are now relatively well understood (Fabricius 2005, De'ath and Fabricius *et al.* 2009, Thompson and Dolman 2010, Thompson *et al.* 2010, Uthicke *et al.* 2010), and responses to pesticide exposure have been documented in controlled laboratory experiments (e.g. Negri *et al.* 2005, Magnusson *et al.* 2008). Synergistic effects between pollutants are relatively poorly understood as yet but likely to be significant, and must be considered when assessing the impact of pollutants on ecological processes. For example, chronic exposure to elevated nutrient or sediment concentrations can increase susceptibility to thermal stress and reduce coral survival and recovery (Anthony *et al.* 2007, Hoegh-Guldberg *et al.* 2007a), and exposure of corals to herbicides has been shown to increase sensitivity to thermal stress (Negri *et al.* 2011). The severity of responses is often a combined function of the magnitude and duration of exposure, and further depends on species-specific levels of sensitivity to the pollutant. For example, differences in sensitivity to nutrient enrichment and sedimentation between species of adult coral can lead to changes in community composition (Fabricius *et al.* 2007b). Over time, chronic sub-lethal stress may decrease the resilience of reef organisms to other pollutants or environmental stress (Hughes *et al.* 2003, Wooldridge 2009a, Anthony *et al.* 2011); however, the potential risks from chronic exposure to multiple contaminants remain unclear (van Dam *et al.* 2011).

Increasing carbon emissions into the Earth's atmosphere are having a range of effects on global climate and the chemistry of oceans. These changes have been shown to impact on coral reefs with potential implications for resilience to future disturbances (Figure 1).

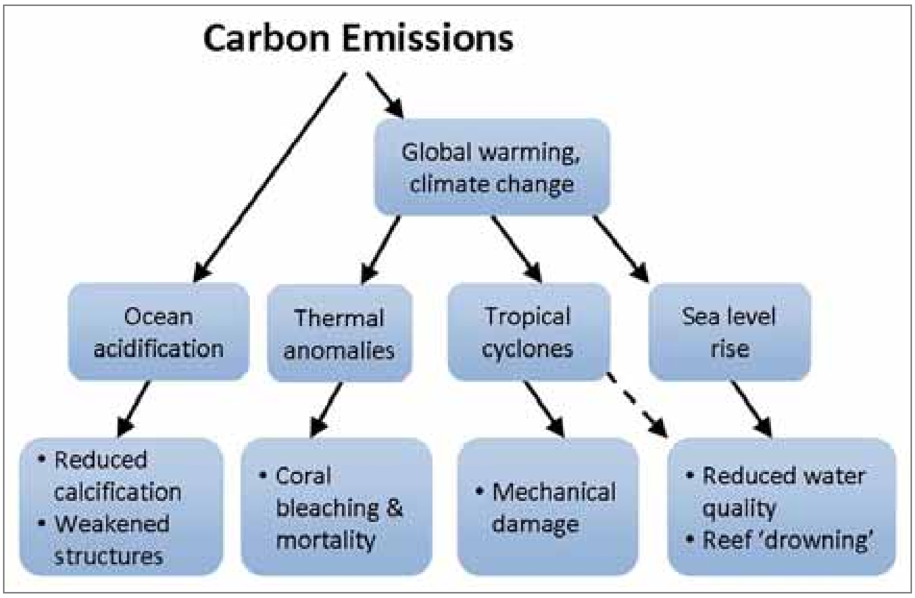


Figure 1: Links between carbon emissions and factors reducing coral reef resilience due to ocean acidification, warming, tropical cyclones and sea level. Adapted from Anthony and Marshall (2009).

This report summarises the key findings of MTSRF research related to the interactions between water quality, ocean warming and ocean acidification in the GBR, and the implications for ecosystem resilience. Sections 2 and 3 provide an overview of the effects of water quality and climate change in the GBR, respectively and Section 4 provides a summary of evidence generated through MTSRF of the links between water quality and resilience to climate and ocean chemistry projections. Finally, conclusions, management applications and future research directions are presented in Section 5.

2. The effects of climate change on the Great Barrier Reef

Projected increases in atmospheric concentrations of carbon dioxide are predicted to increase sea surface temperatures (SSTs) on the GBR by 1-3°C above the present average temperature, while waters will become more acidic, the sea level will continue to rise, and weather events will become more extreme by the year 2100 (Lough 2007, Ramasamy 2010; Table 1). The projected effects of climate change (including ocean acidification) on GBR species, habitats and industries were assessed by the Great Barrier Reef Marine Park Authority (Johnson and Marshall 2007) and formed the foundation for the 'Great Barrier Reef Climate Change Action Plan 2007-2012' (GBRMPA 2007). MTSRF-funded research has aimed to build on these efforts to contribute to the understanding of the impacts and interactions between climate variables, and other stressors on the ecosystem. As identified in the publication 'Climate Change and the Great Barrier Reef' (Johnson and Marshall 2007) and the 'Great Barrier Reef Outlook Report 2009' (GBRMPA 2009), climate change poses a major threat to the survival of coastal habitats, in particular coral reefs, with flow-on effects for other organisms in the GBR. At present, coral bleaching is episodic rather than chronic. However, in the last fifty years, an increase in the baseline temperature of reef waters is narrowing the gap between a regular summer and a coral bleaching season, with a +0.41°C warming observed in the twenty year average (1980-2009) compared to the previous century (1871-1900) (Figure 2). In addition, the decline in ocean pH is likely to have affected calcification rates in marine organisms, particularly corals, and rising sea-levels and increasing incidence of extreme weather events compromise the resilience of reef ecosystems to cope with other stressors (GBRMPA 2009). For example, recent MTSRF-funded research suggests that coral diseases like 'white syndrome' are more prevalent following above average summer SSTs (Hughes 2010).

Coral bleaching occurs in response to physiological stress, and is the result of the symbiotic zooxanthellae (*Symbiodinium*) being expelled by the coral host. For example, sudden changes in salinity, irradiance or temperature can cause corals to bleach (Gleason and Wellington 1993, Glynn and D'Croz 1990, Hoegh-Guldberg and Smith 1989, Jokiel and Coles 1990). Exposure to chemicals such as pesticides may also trigger bleaching, particularly if corals are exposed to poor water quality (Jones and Hoegh-Guldberg 1999, Hoegh-Guldberg *et al.* 2007a, b, van Dam *et al.* 2011). All species of reef-building coral have 'thermal thresholds', the temperature threshold above which bleaching occurs. Thermal thresholds vary spatially and temporally, reflecting adaptation or acclimatisation to local conditions and intrinsic differences in susceptibility to bleaching (Berkelmans and Willis 1999, Baker 2003). Bleaching events can affect individual polyps, an entire colony, or large areas of reef consisting of many different colonies. Bleaching can result in decreased reproductive capacity, increased vulnerability to disease, decreased growth and calcification rates, and ultimately coral death (Done *et al.* 2003, Fabricius *et al.* 2007a). Whether bleaching leads to coral mortality depends on a number of factors. Key findings of MTSRF-funded research (Hoegh-Guldberg and colleagues) have shown that the risk of coral mortality after a bleaching event is related to a combination of several factors, such as the rate of loss of photopigments, duration of the bleaching event, degree of heterotrophy, and the energy reserves of the corals (Hoegh-Guldberg 1999, Eakin *et al.* 2009, Wilkinson and Hodgson 1999).

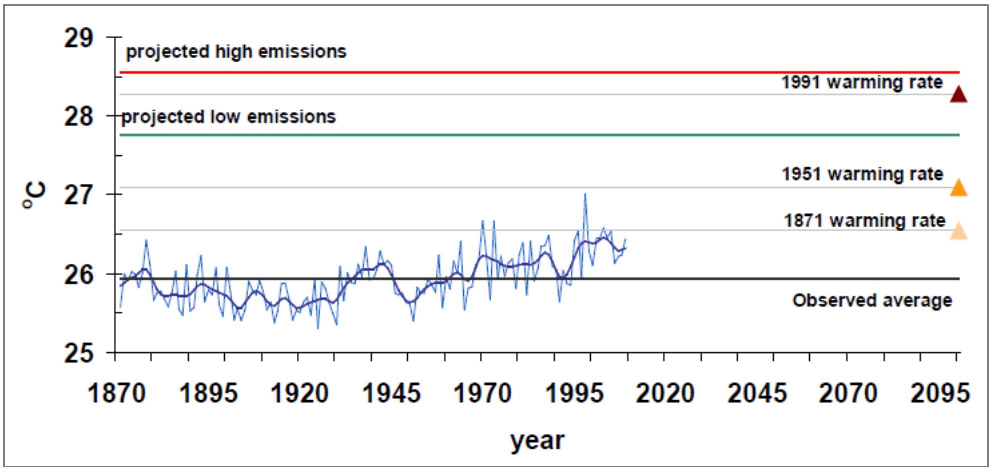


Figure 2: Observed (1871 to 2009) and projected (to 2100 for A2 and B1 emission scenarios) annual sea surface temperatures for the GBR. Central black line is observed average annual SST, 1871 to 2009. Data sources: HadISST and CSIRO. Source: J. Lough, Australian Institute of Marine Science.

Table 1: Climate change projections for the GBR region under low (B1) and high (A2) emissions scenarios. Temperature and rainfall projections are relative to the twenty-year 1980-1999 baseline. Projections represent the range of model predictions (10th to 90th percentile). Data adapted from Lough 2007, Lough *et al.* in press, Ramasamy 2010.

Variable	2030	2070 (B1)	2070 (A2)
Air temperature (°C)	+0.6 to +1.2	+1.1 to +2.0	+2.0 to +4.0
Sea surface temperature (°C)	+0.4 to +1.0	+0.6 to +2.0	+1.0 to +3.0
Annual rainfall (% change)	-8 to +8	-14 to +13	-25 to +25
Sea level (cm)[a]	+13 to +20	+49 to +89[b]	
Ocean pH	-0.1	-0.2	-0.3

[a] Relative to 1990 baseline.

[b] Does not include the melting of the polar ice sheets as significant uncertainty exists regarding melting rate, which will influence the magnitude of sea level rise.

2.1 Coral bleaching susceptibility

Susceptibility and response to bleaching varies geographically between coral colonies, and between and within coral species (Marshall and Baird 2000), indicating that corals may, to some extent, be able to acclimate or adapt to rising SSTs. Indeed, emerging evidence from studies on the molecular basis of resilience suggests that corals with prior exposure to high temperatures are more resilient to future increases in temperature (Done *et al.* 2003, Smith 2008, Hoegh-Guldberg 2010). Hence, variations in bleaching tolerance may be the product of the prior thermal history of the coral, and genetic differences in coral and zooxanthellae physiology (Berkelmans and van Oppen 2006).

In corals, acclimation to an altered thermal regime can occur through a change in the physiology of the host (Coles and Brown 2003), physiology of the symbiont (e.g. photo-protective mechanisms), and a change in the relative proportion of symbiont types within colonies, from endogenous or exogenous pools of symbionts (Baker 2003, Coles and Brown 2003, Anthony and Hoegh-Guldberg 2003). Adaptation, however, can provide a genetic basis for a change in tolerance through differential reproductive success or selective mortality of less thermally tolerant individuals. Major coral-bleaching events are likely to impose strong selective pressures on coral populations and potentially trigger mechanisms of community shifts, adaptation, physiological acclimation or changes in genotype-phenotype interactions in the surviving coral symbioses. For example, thermal tolerance of three coral genera (*Acropora, Pocillopora* and *Porites*) was investigated following the 1998 global mass bleaching event and during the subsequent, more severe event in the GBR in 2002 (Maynard *et al.* 2008a). In 2002, bleaching severity in the three genera was 30-100% lower than predicted from the relationship between severity and thermal stress in 1998, despite higher solar irradiances (Maynard *et al.* 2008a). Interestingly, coral genera most susceptible to thermal stress *(Pocillopora* and *Acropora)* showed the greatest increase in tolerance. It appears that the heating rate and the productivity of symbionts are key factors in determining the flexibility of thermal thresholds in reef-building corals (Hoegh-Guldberg 2010). Thus, even over relatively short time-frames (four years), a major bleaching event can lead to community shifts and possibly enhanced thermal tolerance of surviving coral assemblages without extensive mortality. However, the extent to which acclimation or adaptation is possible in all species is unknown.

Different corals and different symbionts exhibit a range of specificities from ‘generalist’ to ‘specialist’; however, host corals generally appear more specific than symbionts (Baker 2003). Symbionts of scleractinian (reef-building) corals are composed of several clades: A, B, C, D and G, and a host may contain a combination of clades (Tonk *et al.* in press). In Pacific corals, symbionts are predominantly from the C and D clades, with A, B and G being found only occasionally at subtropical or temperate latitudes (Baker 2003). For example, 90% of corals sampled from the GBR contained clade C, 9 % clade D and 1% clades A, B and G. The occurrence of clade D appears to be higher in more northerly locations of the GBR as well as inshore locations within the central Mackay/Capricorn section. The wide range of interactions suggests the symbiosis confer evolutionary and ecological flexibility on both the coral and zooxanthellae.

Molecular studies have shown that the coral host-symbiont partnership can shift phenotypically in response to thermal stress, with the up- and down-regulating of different symbiont clades possible. For example, in the Keppel Islands, a severe bleaching event in 2006 affected approximately 95% of corals, resulting in 40% mortality (Maynard *et al.* 2010, GBRMPA 2008), however by 2009 many of the affected areas had recovered (Hoegh-Guldberg 2010). In the short period of recolonisation following bleaching, it appears that the dominant symbiont changed from the light and heat sensitive clade C2 to the more thermally robust clade C1 and D, which has been shown to occur in laboratory experiments (van Oppen *et al.* 2005, Berkelmans and van Oppen 2006). However, when SSTs returned to average values, the clade C1 and D were down-regulated and the clade C2 was up-regulated. Studies have suggested that this is due to lower growth and reproductive rates in corals with clade C1 and D (Little *et al.* 2004); hence, in favourable conditions the clade C2 provides corals with a competitive advantage. Clade D appears to be an opportunistic taxon that up-regulates in recently stressed or marginal habitats, and/or bleached corals in the process of recovering (Baker 2003, van Oppen 2001). Clade D may therefore be a useful indicator species for inclusion in monitoring programs to detect early signs of stress or to assess the impact of chronic stressors on ecosystem health. Declining water quality and projected increases in SST suggest that clades C1 and D may become more dominant; indeed, changes in symbiont phenotype may be occurring in affected ecosystems such as inshore reefs subject to significant terrestrial impacts (van Oppen 2001).

A number of factors may determine the genetic diversity and connectivity of *Symbiodinium* populations including regional differences in population history and hydrodynamic regimes, the mode of host-symbiont acquisition, and variation in free-living ecology among *Symbiodinium* types. For example, patterns of population structure of clade C symbionts in a widely distributed soft coral reflected longshore circulation patterns and limited cross-shelf mixing, which suggests that passive transport by currents is the primary mechanism of dispersal (Howells *et al.* 2009). In addition, the genetic diversity of clade C symbionts on inshore reefs was lower than on offshore reefs (Figure 3), which may reflect patterns of bleaching or migration (Howells *et al.* 2009). To limit the extent of long-term and irreversible decline in ecosystem health, effective management needs to incorporate the spatial scales at which genetic material is exchanged into management models (Section 5) (Palumbi 2003, Sale 2004, Ridgway and Gates 2006).

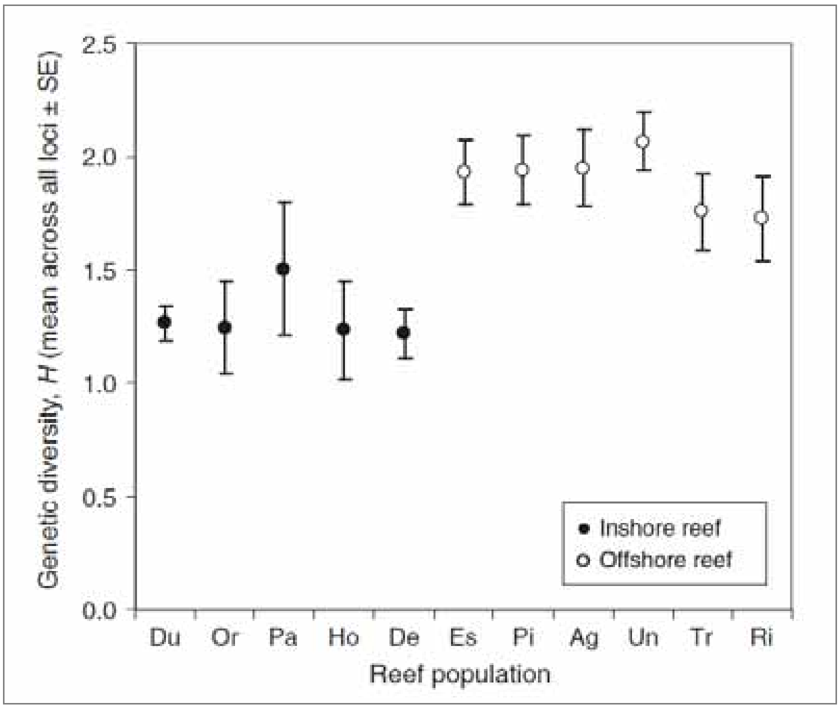


Figure 3: Genetic diversity of clade C symbionts in a widely distributed soft coral on offshore and inshore reefs of the GBR. Genetic diversity is measured by Shannon's Index (H) averaged (+/- SE) across microsatellite loci. Abbreviated reef names: Dungeness Reef (Du), Orpheus Island (Or), Great Palm Island (Pa), Hook Island (Ho), Deloraine Island (De), Escape Reef (Es), Pickersgill Reef (Pi), Agincourt Reef (Ag), Undine Reef (Un), Trunk Reef (Tr), Rib Reef (Ri) (Howells *et al.* 2009).

The spatial resilience of coral reefs may be significantly increased by the diverse and flexible nature of the symbioses. The genotype of symbiotic dinoflagellates appears to be highly diverse and plays an important role in determining environmental tolerance ranges of the coral host (Tonk *et al.* in press). However, the genotype of *Symbiodinium* is not the only determinant of the thermal tolerance of corals. Connectivity within and between coral populations is an important component of reef resilience, because connectivity promotes flexibility in response to environmental variability. Exchange of larvae creates and maintains high levels of genetic diversity, which enables populations to potentially adapt to environmental change and facilitates reseeding of damaged reefs (Reed and Frankham 2003, van Oppen and Gates 2006). In addition, the spread of selectively advantageous alleles involved in physiological responses such as bleaching resistance is a potentially important consequence of larval migration. Furthermore, gene flow increases local effective population sizes, thereby enhancing the resistance of populations to rapid random changes in allele frequencies from one generation to the next through genetic drift. Highly specialised symbionts with low dispersal potential are likely to be susceptible to losses of genetic diversity in the face of increased disturbances unless they are able to adapt, which could impact on the fitness and recruitment success of the host (Howells *et al.* 2009) with flow-on effects for ecosystem health. Furthermore, as the climate becomes more variable and bleaching events become more frequent, research into whether recovery from bleaching events occurs predominantly from exogenous or endogenous sources becomes increasingly relevant. Therefore, the challenge may be how well environmental managers can maintain connectivity between reefs.

Box 1: Predictive models of coral adaptation

Differences in bleaching susceptibility of different symbiont taxa may be applied to predict future incidences and severity of bleaching. To date, the ability to assess the vulnerability of the GBR to increases in SSTs has been limited by a lack of knowledge about the capacity for coral symbioses to acclimate or adapt to thermal stress over ecologically realistic time-frames (Hoegh-Guldberg 2004). What fraction of symbionts is susceptible to bleaching and does community structure of symbionts remain constant over time? The range of bleaching tolerances among corals inhabiting different thermal niches suggests that some coral symbionts have the capacity to adapt to higher SSTs than those currently experienced (Coles and Brown 2003). MTSRF-funded research has investigated whether bleaching sensitivity between geographically distinct, con-specific coral populations growing along a natural thermal gradient was caused by differences in gene expression (reflecting either local adaptation or acclimatisation), and/or by the presence of distinct alleles (due to selection and local adaptation) (Hughes 2010). Experimental results will be used to develop a theoretical model to describe the potential for corals to adapt to climate change and the likelihood of changes in community structure or localised extinctions.

2.2 Coral bleaching mortality risk

Energy acquisition by the coral-algal symbiosis is determined by the rate of photosynthesis and energy transfer by symbionts (Muscatine 1990) and heterotrophy by the host (Anthony and Fabricius 2000). The energy status of corals has implications for survival and fitness, and is a function of total energy intake and loss, and energy allocation between maintenance, growth and reproduction (reviewed by Koiijman 2000). Energy status and reserves, therefore, influence whether bleaching increases the risk of mortality depending on the duration and severity of the event (Anthony *et al.* 2009). Consequently, highly heterotrophic coral species in plankton-rich environments may be less critically affected by loss of symbionts than species that are predominantly phototrophic. In addition, the plasticity of the heterotrophic response influences the risk of mortality, with some species increasing rates of heterotrophy during bleaching and recovery, fully compensating for reductions in photosynthetically acquired carbon (Grottoli *et al.* 2006). Indeed, models showed high rates of heterotrophy and large lipid reserves delay the onset of mortality during bleaching events (Figure 4) and promoted survivorship (Anthony *et al.* 2009).

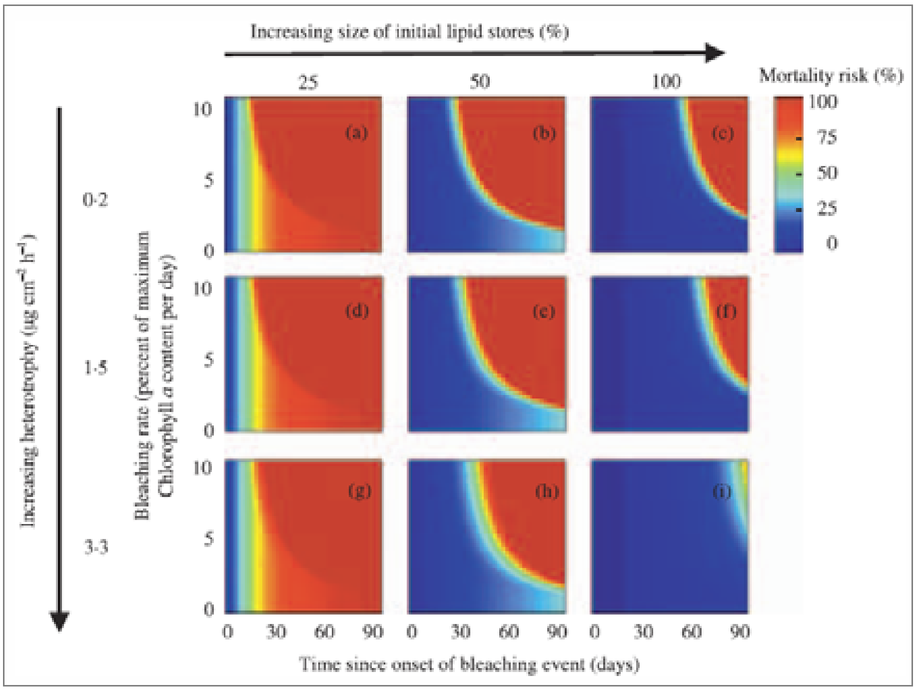


Figure 4: Modelled effects of the size of initial lipid stores before bleaching (top arrow), rate of heterotrophy (left arrow), bleaching rate as a percent of maximum chlorophyll *a* content (*y*-axis), and days since the onset of bleaching for a three-month bleaching event (*x*-axis) on the risk of coral mortality (Anthony *et al.* 2009).

The implications of these results are that corals unable to increase their feeding rates and that are not recolonised by zooxanthellae in the critical period following bleaching will deplete lipid stores to critically low levels, leading to higher mortality. Other stressors that negatively influence lipid stores prior to a bleaching event (e.g. anomalous spring temperatures, Weeks *et al.* 2008; terrestrial pollutants from flood events, Devlin and Brodie 2005) will influence the severity of bleaching. In addition, the availability of resources during a bleaching event may differentially affect coral mortality along eutrophication gradients (e.g. from inshore to offshore reefs), and the timing of the bleaching event relative to seasonal cycles in lipid levels is also likely to be a determinant of coral mortality under future events.

Box 2: Secondary effects of climate change

The cost of not adequately managing for reef resilience could result in increases in the severity and frequency of disease outbreaks with increasing thermal stress. MTSRF research contributed to the development of a predictive tool for outbreaks of 'white syndrome', a serious disease affecting corals of the GBR (Hughes 2010). Average white syndrome abundances were much higher following the exceptionally warm 2001/02 summer than in any other year that reefs have been surveyed (Australian Institute of Marine Science (AIMS) Long-term Monitoring Program). Modelling showed that white syndrome outbreaks require both a high heating rate and high cover of corals in the genus *Acropora*, which is consistent with monitoring data showing 100% incidence of outbreaks at sites with these features. The predictive tool based on the model identifies regions approaching thermal thresholds known to have caused outbreaks of white syndrome in the past. Spatially mapping areas of past and potential future outbreaks of white syndrome will enable researchers to answer the following questions:

1. Is white syndrome induced by pathogens or caused by programmed cell death?
2. What percent of cases result in mortality, and is mortality spatially variable?
3. Is there a correlation between spatial variation in outbreaks and other factors such as host cover and summer max SSTs?

Knowledge in the above areas will assist management to implement actions to influence processes driving the susceptibility to and recovery from white syndrome. The production of a web-based predictive tool for coral diseases compatible with the Google Earth™ platform will inform management and provide an early warning system for coral disease outbreaks. Such a predictive tool has broader applications, and investigations are underway into the transferability to the Pacific and other coral reef regions globally.

2.3 Ocean acidification

Another threat to the GBR ecosystem is the declining pH of the ocean, or ocean acidification, caused by carbon dioxide (CO_2) from the atmosphere dissolving into solution. The world's oceans currently absorb approximately 25% of CO_2 generated by humans, with about 40% of this occurring in the Southern Ocean (Global Carbon Project 2008, Levitus *et al.* 2005, Raven *et al.* 2005). Research shows that ocean acidification decreases the ability of reef-building corals, marine plants and other marine organisms to build calcium carbonate skeletons or shells. A lowering in pH leads to a reduction in the saturation state of aragonite, the chemical building block of reef corals. Projections by MTSRF researchers indicate that ocean acidification may lead to increased levels of severe bleaching, thus becoming a significant concern by mid-century (Hoegh-Guldberg 2010).

Box 3: Linking oceanographic patterns, bleaching events and predictions of coral mortality

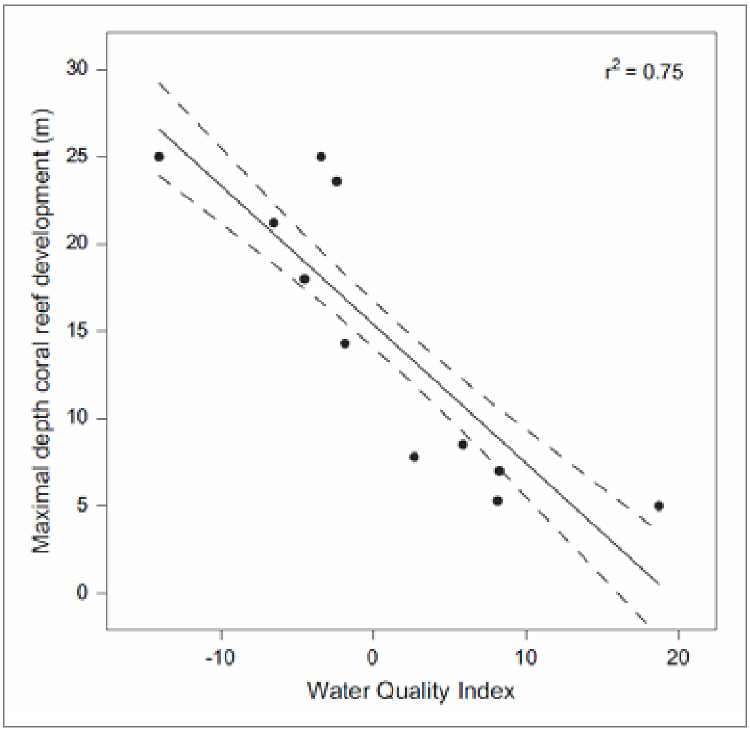
MTSRF-funded research has investigated the influence of mesoscale oceanographic patterns on mass coral bleaching. Remote sensing data of chlorophyll and SST at high resolution (1 km) has revealed distinct 'biogeographic provinces' across the GBR region. The data will be used to investigate how broad-scale climate change phenomena and key environmental variables translate into changes within the GBR ecosystem and impact on reefs at the meso- and local-scale. The spatio-temporal variability in physical dynamics will be linked to biological response to determine whether particular reef systems may be heated or cooled (flushed) and hence the degree of exposure to thermal stress that causes coral bleaching and mortality. This approach was trialed in the southern GBR where local environmental and biological features were used to assess bleaching patterns and resilience at the reef-scale to future thermal events (Maynard *et al.* 2010).

Environmental stress, physiological response and mortality are inter-related (Gurney *et al.* 1996). Models that integrate the effect of key environmental factors on the coral stress response and the extent to which this is influenced by historical adaptation and potential acclimatisation, will improve the ability to predict risk of coral mortality and project future changes in reefs with increasing SSTs. For example, models that build on multiple, nested layers of biological and ecological information have more power to accurately predict sublethal stress, the risk of mortality during and following a bleaching event, and the subsequent probability of population and community recovery (Hoegh-Guldberg 2010). Enhanced predictions of broad-scale mortality would enable management to understand potential shifts in community composition and develop a response strategy to facilitate recovery following bleaching events.

3. The effects of declining water quality on the Great Barrier Reef

Coral reefs usually form in oligotrophic waters (Hoegh-Guldberg 1999, Wilkinson 2004) and natural gradients from coastal to oceanic conditions in temperature, light, nutrients and trophic interactions combine to create a functioning reef ecosystem. Research suggests that cross-shelf gradients in water quality are more persistent in regions with high agricultural run-off than in more remote areas of the GBR (Brodie *et al.* 2007, Furnas *et al.* 1997, Fabricius and De'ath 2004). Changes in water quality are known to influence the physiology, trophic structure and ecology of benthic coral reef assemblages (van Woesik *et al.* 1999, Fabricius 2005, Fabricius *et al.* 2005, De'ath and Fabricius 2010). For example, the maximum depth for reef-building coral growth is closely linked to an index of water quality that increases from turbid to clear-water based on gradients of nutrients, sediment, irradiance and coral reef development (Figure 5; Cooper *et al.* 2007). However, it is often difficult to establish causal relationships between terrestrial run-off and reef health along natural gradients, because many co-occurring contaminants plus reduced salinity interact to affect the ecological balance of the GBR ecosystem.

Figure 5: Relationship between the maximum depth of coral reef development and the water quality index in the Whitsunday Islands, central GBR (Cooper *et al.* 2007).



In a comprehensive review of the effects of terrestrial run-off on the ecology of corals and coral reefs, Fabricius (2005) identified four main factors driving responses to changing coastal water quality: (i) increased dissolved inorganic nutrients, (ii) increased particulate organic matter, (iii) decreased light availability due to turbidity, and (iv) increased sedimentation. The review evaluated the effects of these factors on important ecological processes such as the growth and survival of hard corals, coral reproduction and recruitment, and organisms that interact with corals (Table 2). Building on this review, recent MTSRF-funded studies focus on the effect of run-off on the growth, survival, reproduction and recruitment processes of reef-building corals (see the companion report by Devlin and Waterhouse (2011) for a more detailed overview of these studies).

Table 2: Synthesis of direct effects of the four major components of terrestrial run-off on adult corals. The arrows indicate the relative strength and direction of the response; a dash indicates that a response is unlikely; POM = particulate organic matter; DIN = dissolved inorganic nitrogen; DIP = dissolved inorganic phosphorus (Fabricius 2005).

	DIN	DIP	POM	Light reduction	Sedimentation
Calcification	↓	↓	↑	⬇	⬇
Tissue thickness	–	–	↑	↓	⬇
Zooxanthellae density	⬆	–	↑	↑	↓
Photosynthesis	↑	↑	↑	⬇	⬇
Adult colony survival	–	–	↑	↓	⬇

3.1 Pollutant sources

Catchments adjacent to the GBR have undergone extensive modification over the past 150 years (Furnas 2003). This has led to the receiving waters of the GBR experiencing a six- to nine-fold increase in the input of nutrients (depending on the nutrient species, Kroon *et al.* 2009), a five-fold increase in the amount of sediment (Kroon *et al.* 2009) and an increase in the zone of influence of nutrient enrichment (Wooldridge *et al.* 2006, Devlin *et al.* 2010). The delivery of contaminant loads to the GBR lagoon is typically episodic and dependent on factors such as fertiliser and pesticide application, geography, geology, and spatial and temporal changes in climate and rainfall across the catchments. The outputs of individual rivers vary, from multiple discharges each year (e.g. Tully River), to one major annual discharge event (e.g. Herbert and Pioneer Rivers), to one major flow every four to ten years (e.g. Burdekin River) (Furnas 2003). The extent of area affected by run-off is greatest for large rivers with intermittent flows such as the Fitzroy, Burdekin and Herbert Rivers, although rivers that have frequent discharges (e.g. Tully River) have a chronic influence on coastal ecosystems (e.g. Devlin *et al.* 2001, Packett 2007, Schaffelke *et al.* 2008). Most of the terrestrial run-off affecting inshore reefs occurs during episodic flood events, predominately during the monsoonal wet season between December and May (Devlin and Brodie 2005).

Recent research has enabled clearer identification of the sources of land-based sediment and nutrients. Many of these findings are summarised in the companion report by Waterhouse and Brodie (2011). Establishing relationships between pesticide use and run-off is not as advanced due to challenges in characterising site-specific transport processes and lack of usage data (see the companion report ‘Review of MTSRF research advancing our understanding of the source, transport and impacts of pesticides on the Great Barrier Reef and associated ecological systems’, by Devlin and Lewis, 2011). There is a strong relationship between fertilised land uses (e.g. sugarcane, banana) and loss of nitrate, nitrite and sometimes ammonia in the wet tropical catchments (Mitchell *et al.* 2006, Faithful *et al.* 2006, Rhode *et al.* 2006, Bainbridge *et al.* 2009b) and Fitzroy basin (Packett *et al.* 2009). In contrast, sediment loss is not as high due to improved soil management measures in place for over a decade in areas of sugarcane cultivation, such as green cane harvesting, trash blanketing and reduced tillage (Bainbridge *et al.* 2006b, Rhode *et al.* 2006, Faithful *et al.* 2006). However, large sediment losses are associated with rangeland beef grazing lands (Brodie and Waterhouse 2009, Bainbridge *et al.* 2006a) despite efforts to reduce erosion by improving vegetation cover (Coughlin *et al.* 2007). Hence, the primary pollutants of concern differ between wet and dry catchments, depending on catchment land uses.

Given the high degree of spatial and temporal variability of the GBR and its catchments, modelling is essential in the interpretation of short-term empirical monitoring data. The SedNet and ANNEX model group has been used at the catchment and sub-catchment scale to predict sediment and nutrient generation, transport and delivery to the GBR lagoon (e.g. Brodie *et al.* 2003, Cogle *et al.* 2006, Armour *et al.* 2007, Kinsey-Henderson *et al.* 2007). Once the sources of key pollutants are identified, pollutant loads can be accurately quantified and as indicated above, have been progressed through MTSRF research (see Waterhouse and Brodie 2010 for a summary; Brodie *et al.* 2009a, b, Brodie and Waterhouse 2010, Kroon *et al.* 2009).

MTSRF-funded research has used innovative techniques to identify sources of sediment from GBR sub-catchments through the analysis of coral cores from inner and mid-shelf reefs using laser ablation analyses (Figure 6). Further, the contribution of different catchment erosion processes to sediment loads was quantified using analyses of trace elements and isotopes (Brodie *et al.* 2009a). This is combined with information on the type and quantity of sediment produced by gully, scald, riverbank, hillslope and sheet erosion processes to assist in identifying management practice changes that will be most effective in reducing sediment loads. Further information on MTSRF research on the cost effectiveness of management practices in sugar cane, bananas and grazing (undertaken by van Grieken and others) is summarised in the companion report ‘Managing for GBR resilience: Socio-economic influences’ by Johnson and Martin (2011).

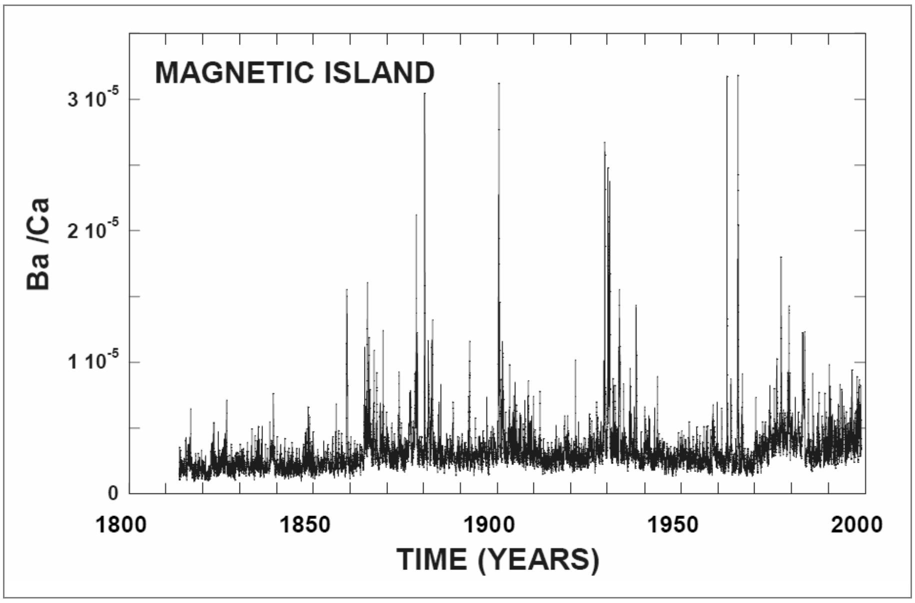


Figure 6: Analysis of a coral core from Magnetic Island showing increased bulk sediment and topsoil erosion in the Burdekin River since European settlement from the 1850s (Lough, 2007).

3.2 Exposure to pollutants

Inshore reefs vary considerably in their resistance to the negative effects of pollutants in run-off and their tolerance of future exposure (e.g. Thompson *et al.* 2011). The exposure (concentration and duration) of the GBR ecosystem to pollutants depends on many factors, including the (i) downstream distance from major discharge sources, (ii) mean annual pollutant load, and (iii) dispersal of flood plumes within the GBR lagoon (Devlin and Brodie 2005, Devlin *et al.* 2010). Concentrations of contaminants in flood plumes are generally diluted with increasing distance from the river mouth, although chemical and biological processes occurring in the plume may influence patterns of retention and removal of some compounds (Devlin *et al.* 2001, Devlin and Brodie 2005, Rhode *et al*. 2006, Packett 2007, Lewis *et al.* 2009, Devlin and Schaffelke 2009). Flood plumes tend to disperse northwards along the coastline, under the influence of south-east winds and Coriolis forcing (Wolanski 1994, Devlin *et al.* 2001) with most terrigenous sediment confined to inshore coastal waters (Packett 2007, Wolanski *et al.* 2008, Devlin and Schaffelke 2009, Devlin *et al.* 2010; Figure 7). MTSRF-funded research has focused on identifying the dispersal of fine sediments as vehicles for pollutants and plume extent (see Wolanski *et al.* 2008, Bainbridge *et al.* 2010).

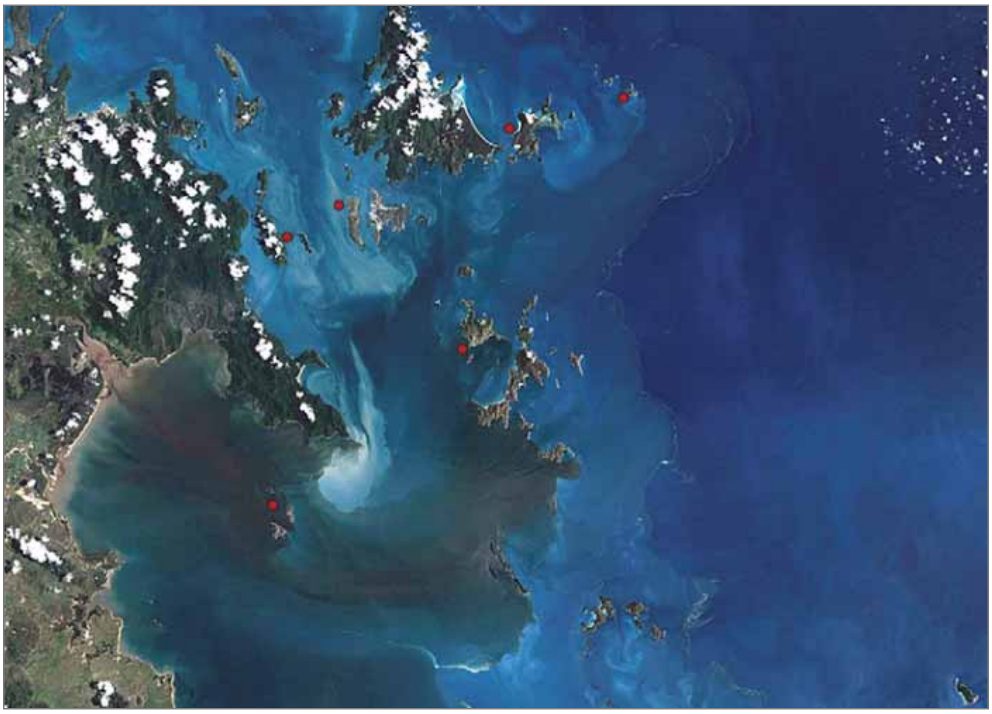


Figure 7: Satellite image (Landsat 5 TM) of the Whitsunday Islands showing a flood plume from the Proserpine and O'Connell Rivers, 28 January 2005. Areas of elevated suspended solids are visible near the river mouths, with areas of increased phytoplankton abundance indicative of nutrient enrichment that extends to the islands. Red circles indicate some of the sampling locations (Cooper *et al.* 2007).

Intensifying human use of the coastal zone and predicted changes to rainfall patterns and storm intensity are likely to increase the variety and quantity of pollutants delivered from the land to the inshore GBR. Inshore ecosystems are most likely to be exposed to pollutants in run-off from small to medium flood events (Fabricius and De'ath 2004, Maughan and Brodie 2009, Brodie and Waterhouse 2009). However, predictions of areas most at risk of degradation are complicated by biological, chemical and physical processes of retention and removal of pollutants, which are not well understood. Cycling of nutrients and local circulation also influence exposure of reefs to pollutants, with reefs in poorly flushed areas, surrounded by shallow seafloor, or subject to frequent disturbances likely to be more exposed and less resilient to pollutant effects (Fabricius *et al.* 2009).

To effectively manage for ecosystem resilience and assess the impacts of improvements in land management, monitoring needs to be able to accurately quantify loads of key pollutants transported by river systems to the GBR lagoon and measure ecosystem response. The companion report by Waterhouse (2010a) provides an overview of optimal techniques developed under the MTSRF Program.

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Box 4: Early warning of high risk bleaching areas

MTSRF-funded research led by Dr Scarla Weeks (University of Queensland) is collaborating with the NASA Ocean Biology Processing Group and members of the US Naval Research Laboratory to generate maps of spatial and temporal variation in euphotic depth for GBR waters. Euphotic depth (Zeu%) is a measure of water clarity (Lee *et al.* 2007) with, for example, Zeu1% reflecting the depth where only 1% of the surface photosynthetic available radiation (PAR) remains. GBR Secchi Depth data have been regressed against the 10% euphotic depth level (Zeu10%), to generate a GBR-validated euphotic depth algorithm. Utilising MODIS satellite data, select monthly mean images of euphotic depth have been generated for GBR waters (Figure 8) however, optically shallow waters require further refinement of the water quality clarity algorithm using GBR-specific *in situ* data. Detecting changes in the transparency of the water column is critical for estimating spatial and temporal variation in exposure of reefs to pollutants that are implicated in coral bleaching events, or are involved in flood dynamics. Coupled with maps of thermal stress reflecting the long-term mean pattern in seasonal variation (Weeks *et al.* 2008, Maynard *et al.* 2008b), data on euphotic depth will provide an early warning assessment of reefs at high risk of bleaching following rapid changes in irradiance, and assist agencies to target management actions that reduce pollutant loads and enhance resilience.

Photic Depth *vs.* *Physical Depth*

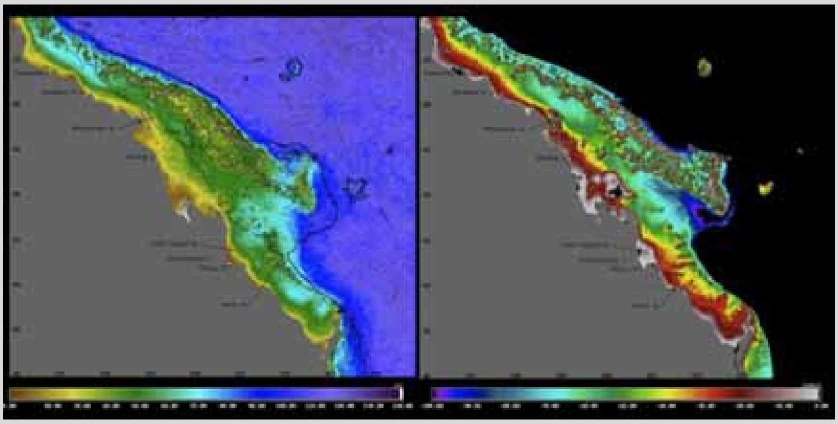

Figure 8: Monthly climatologies of euphotic depth for the southern GBR for (a) March 2007 (left), and (b) September 2002-2009 (right) (Weeks *et al.* 2010).

3.3 Pollutant interactions

The effect of terrestrial run-off on inshore GBR communities depends on the predominant contaminant in the run-off, however interactive effects are likely to be important drivers of community structure. For example, crustose coralline algae are more sensitive to fine sediment when diuron is present and combined exposure to sediment and diuron retarded recovery of photosynthesis (Harrington *et al*. 2005). Data from MTSRF-funded research shows that exposure to the herbicides diuron, atrazine and hexazinone reduces the ability of coral and foraminifera symbionts to photosynthesise and that this effect is more severe in elevated SSTs (Negri *et al*. 2011, van Dam 2011). Combining environmentally relevant concentrations of diuron or atrazine with SSTs above 30°C can also lead to a synergistic damage to photosystem II in coral symbionts (Negri *et al.* 2011). Another recent study showed that protecting coral larvae from copper pollution can significantly improve the ability of coral larvae to settle at elevated SSTs (Negri and Hoogenboom 2011). In addition, a recent study by Humphrey *et al.* (2008) confirmed that environmentally relevant levels of suspended sediments, salinity and dissolved inorganic nutrients interact to suppress fertilisation and development in corals. Interestingly, low concentrations of each contaminant alone had no effect on fertilisation rates, highlighting the complex nature of changing water quality on coral ecology. Moderate levels of particulate organic matter may benefit growth of some coral species (Figure 9). However, at higher levels feeding saturation occurs and the associated reduction in light, sedimentation and dissolved inorganic nutrients negatively affect rates of calcification, photosynthetic compensation points, recruitment success, species richness, community structure, and may result in the proliferation of macroalgae and Crowns-of-thorns starfish (COTS). If coral recruitment is negatively affected, any reduction in fertilisation rates will have profound consequences for the recovery of coral reefs from disturbances (Humphrey *et al.* 2008).

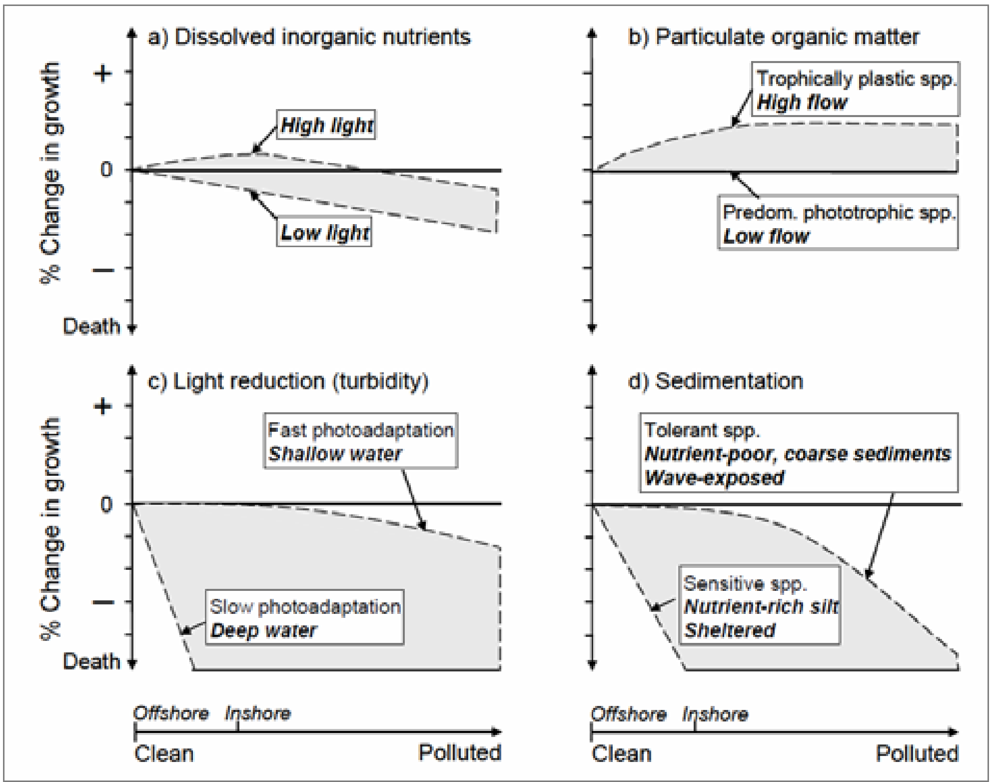

Figure 9: The effect of dissolved inorganic nutrients, particulate organic matter, turbidity, and sedimentation on coral growth (*y*-axis) and survival along a hypothetical environmental gradient from coastal to oceanic conditions (*x*-axis). Grey shaded area approximates species-specific differences in response (normal font) and local environmental conditions (bold italic font) (Humphrey *et al.* 2008).

Similarly, low levels of sediment that alone would not reduce coral recruit survival, kill newly settled recruits when combined with polysaccharides (marine snow) (Fabricius *et al.* 2003). Collectively, these results demonstrate the need to further investigate interactions between contaminants on reefs, especially those adjacent to catchments with a high sediment load, such as the Burdekin River. While some of the specific mechanisms remain unknown, it is clear that the combined effects of terrestrial contaminants will reduce coral reef resilience to increasing frequency of disturbances in a changing climate.

Intrinsic differences in the sensitivity and susceptibility of inshore reefs to the effects of terrestrial run-off may underpin spatial variation in resistance and resilience to climate change (Cloern 2001), together with physical processes governing local exposure to contaminants (Devlin *et al.* 2003, Devlin and Brodie 2005, Fabricius *et al*. 2007b). Increases in SSTs can have acute, chronic, sub-lethal or lethal effects on species with narrow temperature ranges, and different species within an ecosystem can respond in subtly different ways to changes in water quality (Anthony *et al.* 2007). The health of GBR ecosystems depends on the integrity of ecological processes, rather than the health or abundance of individual species or groups of species. Measures of ecosystem health, therefore, require an approach that provides a quantifiable measure of the integrity of ecological processes (Sheaves *et al.* 2007). Assessing the effects of climate change on ecological processes is complicated by the multitude of possible interactions between reef organisms. The impact of bleaching on productivity and species diversity is not well known. Variation in SSTs, ocean currents and upwelling events may individually or collectively adversely affect key ecological processes that flow on to affect the whole ecosystem.

4. Linking water quality and resilience to climate change

Considerable MTSRF research effort has been directed towards improving understanding of the interactions between climate change and declining water quality on the GBR ecosystem in order to generate practical management options that can be used to build and maintain resilience. Under present carbon emissions projections, increasing SSTs will shift the frequency of bleaching events from episodic to annual, storms will become more intense, and ocean acidification will undermine calcification and reef structures. The persistence of hard coral dominated reef-scapes beyond 2050 will therefore depend on their ability to maintain essential ecosystem functions (Hoegh-Guldberg *et al.* 2007a, Veron *et al.* 2009).

Even in areas of relatively good water quality, field data suggest that an average recovery interval of greater than five years is required between major disturbance events to maintain present reef conditions (Wakeford *et al.* 2008). However, in the last decade, the nature and heightened frequency of major disturbances such as coral bleaching, COTS outbreaks and severe storms have exceeded the capacity of many reefs around the world to recover (Wilkinson 2004). For example, in 1998, more than 16% of the world's tropical coral reefs were seriously degraded by a global bleaching event, with 50-90% mortality in some regions (Wilkinson 2004). The GBR fared comparatively well, which has been attributed to greater reef resilience from lower population pressure, targeted ecosystem management and, relatively good water quality (GBRMPA 2009). However, more significant thermal events in 2002 and 2006 in the GBR and southern GBR respectively, resulted in greater coral mortality and highlighted the increasing frequency (or reduced return interval) of these disturbances.

Given that water quality continues to decline and the overall outlook for the GBR is 'poor' (GBRMPA 2009), it is likely that the additional pressure of climate change will result in a shift in reef productivity, community structure, ecosystem function, and the provision of ecosystem goods and services. The situation facing management agencies is the problem of how to minimise the magnitude of this shift by managing for resilience, through maintaining and improving water quality, improvements in land management and coastal development. As a consequence, considerable MTSRF-funded effort has focused on research projects aimed at increasing understanding of the spatial and temporal connections between pollutant run-off and the thermal bleaching susceptibility and mortality of corals.

4.1 Water quality as a driver of resilience to climate change

The complex, and often seemingly contradictory, effects of climate change and eutrophication on coral reefs reveals the challenges of managing for ecosystem resilience. On one hand, even though the extent to which corals are able to acclimate or adapt to temperature perturbations is unclear, there is evidence that prior exposure to extreme temperatures enhances the resilience of inshore corals to future exposures (Section 2.1). Further, the risk of coral mortality during a bleaching event and subsequent recovery phase decreases with increasing energy reserves and degree of plasticity in heterotrophic response (Section 2.2). Dissolved inorganic nutrients such as nitrogen, which are rapidly converted into particulate organic matter within hours or days, may have a beneficial effect on the growth and energy reserves of heterotrophic coral species at moderate levels (Section 2.2). Heterotrophy may account for spatial differences in bleaching-induced stress and mortality between reefs with varying trophic resources. Hence, it would appear that reefs with previous exposure to high temperatures and terrestrial run-off can be more resistant and resilient to climate-change related disturbance in the short term.

However, while specific communities of coral are likely to incur some benefit from increased nutrient availability, the interactive effects of contaminants in run-off and increases in SSTs on reef heath are overwhelmingly negative. For example, MTSRF-funded research (Negri *et* *al.* 2011) showed elevated SSTs enhanced the negative effect of low concentrations of three commonly detected photosystem-II herbicides on photosynthetic efficiency in corals (additive effect), with a greater effect on chronic photoinhibition (synergistic effect) (Negri *et al.* 2011). At high levels of particulate organic matter, the associated reduction in light negatively affects rates of calcification, photosynthetic compensation points, and community structure and function in corals, and increases susceptibility to bleaching (Section 2.1). Hence, exposure to terrestrial run-off can increase the sensitivity of reef ecosystems to elevated SSTs that have already been experienced in GBR areas during previous summers.

Further experiments investigating interactions between water quality and climate change suggest that for at least one foram species, additive effects of the two stressors exist, with foram growth rates negatively affected by increased nutrient concentrations and increased temperature (Uthicke *et al*. 2010). At temperatures greater than 32°C, herbicides had a lethal effect on diatom-bearing species of foraminifera (van Dam *et al.* 2011). Foram species were also tested for susceptibility to thermal stress, and showed responses at temperatures only slightly above current summer maxima, allowing identification of thermal thresholds for several foraminiferan species. The development of a FORAM index provides an effective bioindicator for the assessment of turbidity and light regimes and organic enrichment of sediments on coral reefs (Uthicke *et al.* 2010).

Understanding properties of reefs or regions that contribute to their resilience could generate practical management options, for example by enabling prioritisation of protection of reefs that have the greatest chance of withstanding degradation, or targeting management actions to increase the resilience of ecologically significant vulnerable areas. For example, the colour darkness of coral communities increases along a gradient from offshore to coastal waters in response to elevated concentrations of nitrate, particulate nutrients and suspended sediments (Figure 10) (Fabricius 2005). The darker pigmentation of coastal corals results in greater thermal absorption and therefore greater sensitivity to thermal stress under conditions of high temperature and irradiance, compared to lighter offshore corals. However, water flow alleviates surface warming in corals, such that high flow reef environments will have a lower likelihood of bleaching than reefs or embayments under low flow conditions (Nakamura and van Woesik 2001, West and Salm 2003).

Figure 10: Coral colour changes substantially along a gradient from coastal to offshore regions of the GBR, with corals significantly darker in coastal waters (Fabricius *et al.* 2007b).

In addition, colony geometry and tissue characteristics of corals may, to some extent, modulate internal light environments and alleviate light stress. Experiments showed the surface temperature of darker corals in turbid water was significantly greater than that of paler corals in clear water environments at comparable sea temperatures, light and flow conditions, and concluded that surface warming of darkly pigmented corals in coastal environments is sufficiently high to exceed bleaching thresholds (Fabricius *et al.* 2007b). Hence a better understanding of the interactions between factors such as flow, irradiance, pigmentation, colony geometry and water quality on the microenvironment of corals may assist in interpreting some of the spatial and temporal variability in coral bleaching across the GBR, and improve identification of vulnerable reefs (Fabricius *et al.* 2007a).

Recent modelling efforts funded by the MTSRF have focussed on linking regional water quality and climate change variables to provide a framework for management agencies to target investments under Reef Rescue to improve reef resilience (Wooldridge 2010). However, the complex interactions between contaminants co-occurring in run-off complicate predictions of spatial and temporal resilience to climate change, and consequently models have generally focussed on only one water quality variable (e.g. sediment or nutrients; Section 3.1). These models, whilst presenting a simplistic assessment of water quality factors underpinning reef resilience, enable management to assess the potential effect of various land-use scenarios or changes in land management practices on water quality. With further development to include factors such as synergistic effects of stressors (Section 3.3), non-linearity of responses (Section 2.2), species-specific thermal thresholds (Section 2.1), spatial differences in thermal histories of reefs (Box 3), community composition (Section 2.3), the effect of pulsed versus chronic exposure to contaminants (Section 3.2), and remote sensing data of spatial variation in physical and hydrodymamic characteristics of reefs (Box 4), the capacity of models to identify specific effects of changes in water quality on reef resilience will improve.

4.2 Modelling resilience of inshore reefs

Corals flourish in clear, nutrient-limited water; hence studies of eutrophication effects on corals have often focused on enrichment with dissolved inorganic nutrients (e.g. Stambler *et al.* 1994, Dubinsky and Stambler 1996, Koop *et al.* 2001, Szmant 2002). MTSRF-funded researchers tested the theory that dissolved inorganic nitrogen (DIN) is an important driver of coral response to elevated temperature (Wooldridge 2009a, b). This theory was postulated on there being direct effects of DIN on coral physiology at high concentrations in poorly flushed locations, and indirect effects from incorporation into benthic food webs and conversion into particulate form (Alongi and McKinnon 2005, Fabricius 2005). Although there is currently no direct experimental evidence of the effect of DIN on thermal sensitivity of corals, the model provides a mechanism for testing interactions between water quality and coral bleaching, and can eventually be used with empirical data inputs.

Wooldridge (2009a) modelled the possible future risk of coral bleaching at inshore reefs located within the region influenced by flood plumes from the Burdekin River under a range of land management scenarios. The exposure of inshore reefs to high concentrations of DIN (using chlorophyll *a* as a proxy) was predicted to lower bleaching thresholds of corals compared to reefs in waters with lower exposure, although the synergistic effects of suspended sediment and associated reductions in light were not considered. The study identified land management actions that lower the delivery of DIN to the reef as potentially enhancing reef resilience to elevated SSTs (Wooldridge 2009b) (Figure 11). Similarly, in the Tully-Murray catchment, it was proposed that a reduction in DIN outputs may influence bleaching-induced mortality compared to no reduction in DIN (Wooldridge 2008).

Further simulations of projected increases in thermal tolerance of corals under a range of climate change scenarios proposed that integrating reductions in DIN with the stabilisation of atmospheric CO_2 below 450 ppm would assist in maintaining coral dominated reefs beyond 2100 (Wooldridge 2010). At present, rates of adaptation are unlikely to be sufficiently fast for corals to acquire increased resistance to increasing SSTs and declining pH even under modest climate change scenarios (Wooldridge and Done 2009). Further research to elucidate the effect of improved land management practices on thermal sensitivity and building and maintaining coral reef resilience are imperative (Bellwood *et al.* 2004, Diaz-Pulido *et al.* 2009, Hughes *et al.* 2010, Marshall and Schuttenberg 2006, Marshall and Johnson 2007).

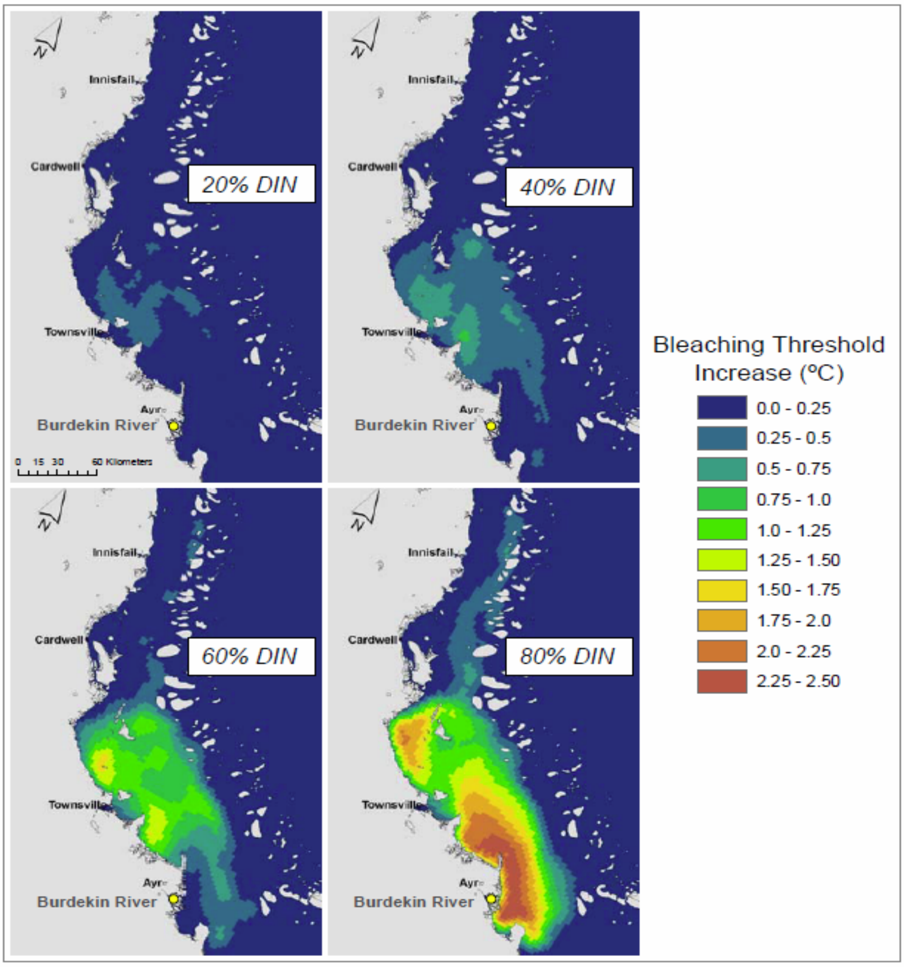

Figure 11: Modelled output postulating the interaction between DIN delivery and bleaching thresholds of corals (Wooldridge 2009b).

4.3 Managing water quality for resilience

MTSRF-funded research has made significant progress in linking research on the delivery and impacts of pollutants on inshore GBR ecosystems and the resilience of reef communities to changes in SSTs and other climate-related disturbances (Sections 2 and 3). This knowledge is critical to inform the design and prioritisation of effective management strategies that aim to enhance the resilience of the GBR to future climate change.

The form of nutrients in run-off has implications for management actions targeting sources of pollutants. For example, floodwaters tend to contain higher concentrations of dissolved organic nitrogen (DON) compared to in-river concentrations, while river waters tend to contain higher concentrations of dissolved inorganic nitrogen (DIN). Using the results of MTSRF modelling that postulated that reducing DIN delivery may improve coral resistant to thermal stress (Wooldridge 2009b) allows for management to consider the minimisation of DIN through improved agriculture practices, such as N-replacement approaches (Wallace *et al.* 2009a).

Soils of fine texture have a high surface area and hence a greater sorption area for pollutants providing a major transport mechanism for phosphorus and other elements to reach the GBR (McCulloch *et al.* 2003, Packett *et al.* 2009, Bainbridge *et al.* 2010). Therefore, reducing fine texture soils will reduce the impact of sedimentation as well as nutrients on inshore ecosystems, with the potential to increase their resilience to thermal stress (Fabricius *et al.* 2009). In addition, quantifying the link between water quality and bleaching risk, together with a spatially explicit model exploring patterns of land use and changes in land management across a range of industries (Henderson and Kroon 2009), will enable management to target areas to focus effort and cost-effective solutions to achieving water quality targets.

Predicting changes in the state of reef communities in response to climate change is complicated by uncertainty concerning the processes driving post-disturbance recovery of coral reefs operating under different levels of environmental and ecological degradation. Uncertainty concerning the potential for the coral-symbiont partnership to adjust or adapt to warming conditions, and uncertainty about the cumulative effects of other stressors on ecosystem health. Managing a complex natural resource such as the GBR requires an understanding of the different pressures on the system and their interactions, as there are likely to be more than one ‘driver’ of reef resilience. Overly simplistic modelling of links between water quality variables and reef resilience may produce negative flow-on effects (maladaptation) that require additional strategies to manage. It is important to quantify how these sources of uncertainty affect the accuracy of predicting shifts in ecosystem structure and function with climate change, especially when comparing alternative policies and management interventions for achieving desired system water quality outcomes.

5. Conclusions and management applications

5.1 Conclusions

In summary, the interactive effects between water quality and climate change has implications for the susceptibility of corals to thermal stress (i.e. bleaching), disease virulence, the risk of widespread coral mortality, and recovery rates post-disturbance. Effective management for GBR resilience requires integration across disciplines and an understanding of the complexity of factors that influence spatial and temporal differences in vulnerability to climate change. For example, resilience to bleaching may vary depending on a reef's thermal history, the frequency and nature of prior disturbances, and its exposure, sensitivity and resilience to pollutants. To date, the development of models that predict bleaching likelihood on the basis of one environmental variable represent a step forward in understanding the complex mechanisms underpinning bleaching susceptibility of reefs to increasing SSTs. With further development, such models can potentially simplify the complex array of biological and physical interactions to deliver solutions to environmental managers seeking to protect and conserve the biodiversity of the GBR.

The information generated through MTSRF has also contributed to an emerging community consensus that water quality is one of the high-priority management issues for the GBR, particularly in the face of climate change. Further empirical data is needed to quantify this relationship and to identify specific actions that will sustain coral reef resilience (Hughes *et al*. 2010). Despite improvements in land management practices through government initiatives such as the Reef Water Quality Protection Plan (Reef Plan; DPC 2009) and Reef Rescue (Australian Government 2007), a long time-lag is expected before any improvement in GBR water quality will translate into measureable changes in the ecosystem (Bainbridge *et al.* 2009a).

5.2 Management applications

Achieving sustainable loads of pollutants and developing mixtures of effective policy instruments and governance arrangements for managing water quality and climate change remain significant challenges at the catchment, regional and GBR scales. It is not known whether current management and policy efforts will be sufficient to achieve water quality improvement that will protect and enhance the resilience of the GBR ecosystem. Part of this uncertainty relates to the fact that climate projections for the region have their own uncertainty making most measures of resilience relative. Improving regional climate projections and the understanding of the links between water quality, ecosystem resilience and future change will assist management. Researchers funded by the MTSRF have made significant progress in furthering understanding of the complex processes and interactions between water quality and resilience to climate change, and developing models that can deal with this complexity, and provide predictive tools for management.

The advanced models developed through the MTSRF can link changes in reef resilience with changes in land management practices, and can provide management with spatial and temporal mapping of exposure to pollutants in run-off, overlayed with areas where poor water quality influences resilience (or lack of) to thermal stress. By showing that coral mortality is related to bleaching severity, thermal event duration, and size of initial energy reserves, reef areas that are most vulnerable to future thermal events can be identified and prioritised for management. Conversely, by using MTSRF-funded results showing that corals that survived prior exposure to high SSTs are more resilient to future thermal events through mechanisms of tolerance and adaptation, management can make informed decisions to protect or enhance this natural resilience as a seed bank for future recovery of less resilience reefs.

MTSRF models that can provide predictions of coral mortality following a bleaching event or future implications of ocean acidification will improve understanding of community composition shifts, which will be important information for monitoring community changes due to climate and again identifying suitable reef areas for protection. This information also has applications for industries that depend on the GBR, in particular tourism operators who rely on healthy coral communities. Being able to predict how reef communities might change under increasing SSTs or ocean acidification can inform future planning for reef-dependent industries. Similarly, MTSRF research that has identified a strong link between temperature anomalies and coral disease and developed a predictive tool linking thermal stress and increased severity and frequency of coral disease, can be used to understand future spatial patterns of coral disease and inform management and industry planning decisions.

Ultimately, GBR ecosystem resilience to future climate change will depend on the rate and magnitude of change, and the effectiveness of improvements in water quality, which has been shown to significantly undermine reef resilience. MTSRF research has provided improved conceptual and empirical understanding of the effects of water quality variables on changes in coral reef community structure, an increased understanding of the negative interactive effects of different contaminants and climate change on ecosystem health, and improved knowledge of the effects of water quality on the resilience of reef organisms to climate change. Collectively, this information can be applied to strengthen existing water quality improvement initiatives in the GBR catchment, provide information on which ecosystems are most at risk and therefore require management focus, as well as inform decisions to prioritise protection of specific reefs and locations as refugia to facilitate future recovery of reefs in the face of climate change.

5.3 Future directions

Scientific information that informs effective management of the GBR must take a holistic approach that considers interactions between multiple stressors, co-occurring environmental factors, and the flow-on effects for ecosystem health. There is an urgent need to integrate monitoring, modelling and experimental knowledge from catchments to the GBR, and to experimentally define the direct effects of water quality parameters on thermal sensitivity, bleaching threshold and recovery after disturbance. In addition, the mechanism by which water quality influences calcification also needs to be understood to predict how the structure of coral reefs may change under future ocean acidification. Ultimately, a single model that predicts spatial and temporal vulnerability to the combined effects of ocean warming, ocean acidification and climate change is critical. Such an integrated model will facilitate the implementation of targeted management strategies by linking GBR ecosystem health with water quality in the lagoon and catchments.

The development of new management practices and the modelling of future changes in land use on water quality remain critical requirements to effectively target resources for water quality improvement in a changing climate. Understanding the transport and trapping of contaminants from the ‘source-to-sea’ has improved greatly however, current monitoring and modelling methods significantly underestimate pollutant loads, and there is likely to be a considerable lag in detection of any effect of changes in land use or land management practices on marine water quality (Bainbridge *et al.* 2009, Bainbridge and Brodie 2009, Brodie *et al.* 2009, Wallace *et al.* 2009b). Hence, improvements under the Reef Plan and Reef Rescue initiative are likely to take years to detect change in ecosystem function. The development of a single model linking transport process across the land-sea continuum is required to provide a framework for linking land management to changes in water quality and ecosystem health (see Webster *et al.* 2008)

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