2013 Scientific Consensus Statement

Chapter 1
Marine and coastal ecosystem impacts

Britta Schaffelke, Ken Anthony, Julieanne Blake, Jon Brodie,
Catherine Collier, Michelle Devlin, Katharina Fabricius,
Katherine Martin, Len McKenzie, Andrew Negri, Mike Ronan,
Angus Thompson, Michael Warne

Image credits: Queensland Government Department of Environment and Heritage Protection, Tourism Queensland.

This document was prepared by an independent panel of scientists with expertise in Great Barrier Reef water quality. This document does not represent government policy.

Table of Contents

Executive summary

The Great Barrier Reef marine ecosystems and their associated catchments are part of a dynamic, interconnected system. Activities within the catchments affect the condition of Great Barrier Reef coral reefs and seagrass meadows, which has declined severely in the period since 2008 (last Reef Plan Scientific Consensus Statement). The understanding of the causes, nature and trajectory of these declines has become much clearer, especially for coral reef systems. The scientific consensus presented in this review confirms (i) that marine water quality continues to be negatively affected by the discharge of excess nutrients, fine sediments and pesticides from the adjacent catchments and (ii) that poor marine water quality is a major cause for the currently poor state of many of the key marine ecosystems of the Great Barrier Reef.

Condition of Great Barrier Reef coral reef ecosystems and response to water quality

1. Large-scale and chronic disturbances have reduced the average coral cover on reefs in the central and southern Great Barrier Reef to less than 50 per cent of what it was in 1985, while the northern Great Barrier Reef has remained stable.
2. Tropical cyclones and population outbreaks of the coral-eating crown-of-thorns starfish (COTS) are the main causes of the observed coral cover declines, and evidence of the link of crown-of-thorns starfish outbreaks to water quality has been greatly strengthened.
3. Some inshore reefs have undergone dramatic changes in coral and foraminifera composition and abundance over the last century associated with multiple disturbances including increased terrestrial runoff.
4. In inshore waters, pulses of suspended sediment input influence levels of turbidity. Runoff can episodically and chronically reduce the levels of light reaching corals.
5. Inshore coral reef ecosystems are directly and negatively affected by increased turbidity and sedimentation of fine particles and organic rich flocs (muddy marine snow), especially after extreme weather events.
6. Early life-history stages of corals are particularly susceptible to sedimentation. Following high river flows and associated increases in sediment and nutrient loads, the densities of juvenile corals on reefs prone to sediment accumulation have markedly declined.
7. Corals in nutrient-enriched or turbid waters appear more vulnerable to temperature stress than those in low nutrient environments.
8. While it seems unlikely that pesticides alone pose a major threat to the entire Great Barrier Reef, the connectivity of catchment, estuarine and inshore Great Barrier Reef ecosystems, the high sensitivity of foundation species to pesticides and emerging evidence that pesticides increase the vulnerability of Great Barrier Reef species to the negative effects of climate change leads us to conclude that efforts to significantly reduce pesticide contamination of the Great Barrier Reef should continue.

Condition of Great Barrier Reef seagrass ecosystems and response to water quality

1. Inshore intertidal seagrass meadows along the Great Barrier Reef developed coast have been declining over the last three to five years and are currently in a poor condition, based on assessments of abundance, meadow size, reproductive effort and epiphyte load.
2. Light is the most critical variable affecting seagrass growth and survival. Pulses of suspended sediments from river runoff increases turbidity and this in turn reduces the levels of light reaching seagrass.
3. In large scale flood events seagrass health can be directly affected by sediment deposition close to river mouths.

4. Although current nutrient concentrations in the Great Barrier Reef have not reached their critical levels for seagrasses there are indications of long term increases in seagrass leaf tissue nitrogen concentrations, increasing epiphyte loads and reduced light availability since records began in 2005.
5. Many coastal and inshore seagrass meadows of the Great Barrier Reef are exposed to herbicides at concentrations that adversely affect seagrass productivity but the contribution of herbicides to recent widespread seagrass losses is unknown.

Condition of Great Barrier Reef mangrove ecosystems and response to water quality

1. The responses of mangroves to poor water quality are complex and poorly understood.
2. Increased sedimentation, e.g. due to upstream vegetation clearing or marine dredging activities, has led to expansion of some mangrove stands. However, excessive sedimentation may also bury seedlings and pneumatophores, reduce growth and cause mortality of mangroves.
3. Higher nitrogen availability favours growth of mangrove shoots relative to roots, enhancing productivity under favourable conditions, but increases vulnerability to water stress. The probability of canopy loss and tree mortality increases with nitrogen fertilisation along gradients of increasing soil salinity and aridity.

Condition of freshwater wetlands and response to water quality

1. The area of non-riverine freshwater wetlands remaining in Great Barrier Reef catchments in 2009 equates to 78 per cent of the pre-European extent (280,000 hectares). Seventy-seven per cent (262,000 hectares) of palustrine (swamps) and 97 per cent (21,000 hectares) of lacustrine (lakes) wetlands remained in 2009.
2. Poor land use and management practices affect many of the remaining coastal freshwater wetlands, with inputs of excess sediment and nutrient and certain irrigation techniques affecting wetland structure and function, for example by facilitating weed growth, loss of connectivity, reduced oxygen levels and flow rate.
3. The effects of poor water quality on freshwater wetland ecosystems are not well understood due to limited research. However, recent evidence shows that the Bowling Green Bay Ramsar site (mostly marine wetlands) and the adjacent freshwater wetlands are at high risk of damage due to pesticide loading.

In conclusion, there is strong evidence that improving water quality within the catchments of the Great Barrier Reef will contribute to enhancing the resilience of freshwater, estuarine, mangrove, coral reef and seagrass ecosystems to other disturbances. Existing data suggest that reducing losses of nutrients, sediments and pesticides from the land will ‘buy’ the exposed areas of the Great Barrier Reef some time by partially offsetting increasing stress and damage from climate factors (increasing mean temperatures, frequencies of heat periods, ocean acidification, intensifying storms and rainfall variability). The most significant impact would be achieved by removing the water quality effect that triggers more frequent crown-of-thorns starfish outbreaks. The documented significant losses of Great Barrier Reef coral and seagrass cover and subsequent implications for other biota such as green turtle and dugong over the recent decades show that such action needs to be given high priority.

Introduction

The iconic Great Barrier Reef marine ecosystems are interconnected with freshwater and estuarine ecosystems, freshwater wetlands and the wider landscape of the Great Barrier Reef catchments. The impacts of poor water quality on both marine and freshwater ecosystems can have far reaching impacts such as on the food chain, nutrient cycling, species distribution, abundance, population size, growth, reproduction.

Increasing pressure from human activities and climate change continues to have an adverse impact on the quality of water entering freshwater wetlands in the Great Barrier Reef catchments as well as the lagoon itself, particularly during flood events.

The relationships between land use, catchment and wetlands management, declining water quality and Great Barrier Reef ecosystem health are now better understood and there is well-documented evidence of the negative impacts of terrestrial pollutants on the condition of coastal and inshore Great Barrier Reef ecosystems (see recent review in Brodie *et al.,* 2012).

In this chapter, the condition and trend of key Great Barrier Reef marine and adjacent freshwater ecosystems and assess all key drivers of change and activities leading to pressures and impacts is reviewed. These include the impacts of land runoff, coastal developments, dredging and other disturbances. The authors focus on information since 2008, the year of the last Consensus Statement, and re-evaluate the statements in the Consensus Statement against this new information.

The conclusions of the 2008 Consensus Statement relevant to this chapter were:

1. Water discharged from rivers to the Great Barrier Reef continues to be of poor quality in many locations.
2. Land derived contaminants, including suspended sediments, nutrients and pesticides are present in the Great Barrier Reef at concentrations likely to cause environmental harm.
3. There is strengthened evidence of the causal relationship between water quality and coastal and marine ecosystem health.
4. Climate change and major land use change will have confounding influences on Great Barrier Reef health.

Synthesis process

This chapter has been co-written by a group of scientists, all of whom are active researchers on the effects of water quality on tropical marine ecosystems, and with specific knowledge and a track record of published research about the Great Barrier Reef.

The contributors to this chapter conducted a comprehensive literature review, accessing the latest scientific information available in peer- reviewed scientific journals and technical reports. The review focused on research that was published or at least available in online resources at the time of writing. To achieve the highest level of transparency, information that was unpublished or not readily accessible was not included in the review of current scientific information about the impacts of poor water quality on Great Barrier Reef marine ecosystems.

Previous Consensus Statement findings

The main conclusion in the 2008 Consensus Statement was that '*There is strengthened evidence of the causal relationship between water quality and coastal and marine ecosystem health.'* This conclusion was mainly based on laboratory and field studies; and data syntheses because the size and variability within the Great Barrier Reef ecosystems makes comprehensive measurements of physical, chemical and ecological conditions challenging. Since the statement, however, the data availability has much improved, not least through the Paddock to Reef Integrated Monitoring Program. The state of knowledge in 2008 was that coral reefs and seagrass meadows showed indications of decline in their condition that could be attributed to exposure to land runoff carrying excess loads of nutrients and fine sediments, as well as agricultural pesticides. Also, further support for the links between nutrient enrichment and crown-of-thorns starfish population outbreaks was presented. Water quality effects on coral reefs were mostly studies along gradients of exposure to identify patterns of responses and allow for correlative analyses. Since 2008, there have been many extreme weather events allowing for studies of ecosystem change due to water quality impacts through time and not only along gradients.

Since the 2008 statement, water quality trigger values have been developed for the Great Barrier Reef lagoon, based on empirical relationships between marine water quality and coral reef responses. These can now be used to put field data into context; e.g. from routine surveys under the Reef Rescue Marine Monitoring Program. In 2008 first indications of interactions between multiple and cumulative impacts were recognised; e.g. that there are negative synergistic effects between herbicides and sediments on crustose coralline algae, which are essential for successful coral recruitment. The understanding of interactions between water quality variables and between water quality and climate change drivers had been identified in 2008 as a crucial knowledge gap in the understanding of the downstream effects of catchment runoff. Current research is slowly improving the understanding of these complex relationships and first results are reported in this updated consensus statement.

Another important knowledge gap identified was the scarcity of numerical models that link marine ecosystem health to end of catchment loads. Research currently underway is developing a comprehensive suite of interlinked hydrodynamic and sediment transport models (eReefs project) that will in time deliver a model that can be used to assess the influence of catchment management actions on Great Barrier Reef health in a comprehensive way.

However, our understanding of the response of Great Barrier Reef ecosystems to the combined scale and frequency of disturbances, and, most importantly, the ability to recover from such disturbances when exposed to land runoff is still incomplete.

Impacts of poor water quality on the marine ecosystem - science consensus 2013

Exposure of marine environments to land runoff

Water quality in the Great Barrier Reef (Great Barrier Reef) is influenced by an array of factors including land-based runoff. River runoff transports agricultural pollutants such as nutrients (Devlin and Schaffelke, 2009; Brodie *et al.,* 2012; Cooper *et al.,* 2007; Turner *et al.,* 2012, in press), sediments (Devlin and Schaffelke, 2009; Bainbridge *et al.,* 2012; Turner *et al.,* 2012, in press) and pesticides (predominantly herbicides) into the Great Barrier Reef lagoon (Bainbridge *et al.,* 2009; Brodie *et al.,* 2012; Lewis *et al.,* 2009, 2012a; Shaw *et al.,* 2010; Kennedy *et al.,* 2012a,b; Shaw *et al.,* 2012; Smith *et al.,* 2012). All pollutants move and disperse during flood plumes, however due to variation in transport, uptake process, degradation (biotic and abiotic) and residence times, different constituents have discrete patterns of behaviour as they move through the estuarine plume. The distribution and movement of the individual constituents in a flood plume varies considerably between the regions in the Great Barrier Reef, particularly between Wet and Dry Tropics rivers (Brodie *et al.,* 2012).

Physical oceanographic studies of plumes in the Great Barrier Reef (Wolanski and Jones 1981; Wolanski and van Senden, 1983), modelling studies (King *et al.,* 1997; 2000) and satellite observations (Figure 2a, Brodie *et al.,* 2010; Devlin *et al.,* 2012) show that river plumes are generally constrained close to the coast by Coriolis forces and the prevailing wind regime, limiting impacts on the more offshore reefs. Flood plumes move in response to prevailing weather conditions and can form estuarine mixing zones far away from the coast. Wet season discharges vary enormously between years (e.g., Turner *et al.,* 2012, in press), and in the right conditions can form very large flood plumes extending offshore into the Great Barrier Reef lagoon, sometimes hundreds of kilometres distant from river mouths (Beman *et al.,* 2005; Brodie *et al.,* 2010; Devlin and Schaffelke, 2009). Under northerly wind conditions, plumes are often deflected seaward and reach the mid and outer-shelf of the Great Barrier Reef (Alvarez-Romero *et al.,* 2013; Devlin *et al.,* 2012; Brodie *et al.,* 2010; Figure 2b).

Recent extreme weather with above median flows in many Great Barrier Reef rivers, particularly the large Dry Tropics rivers (Turner *et al.,* 2012, in press), has resulted in large lagoon areas north of the Fitzroy and the Burdekin being inundated on an annual basis (Devlin *et al.,* 2012; Alvarez-Romero *et al.,* 2013; Brando *et al.,* 2011; refer also to Chapter 2). This is supported by the visualisation of movement of low salinity water from the Burdekin using the Great Barrier Reef 4 (3D) hydrodynamic model (Furnas *et al.,* 2013) and the movement of visible surface flood plumes from the Fitzroy River northward (Devlin *et al.,* 2012), which show for example that the flow of Burdekin and Fitzroy rivers can reach the Whitsunday Islands, and that Burdekin flood plumes reach beyond weather stations positioned approximately 40 kilometres north of the Burdekin River mouth.

High flow conditions resulting in the formation of flood plumes would have always moved terrestrially sourced material into the Great Barrier Reef. For example, low salinity conditions would have influenced reef condition over hundreds of years. However, there have been considerable changes in the volume and composition of these river flows. Pesticides have only been transported to the Great Barrier Reef since their use started in catchments adjacent to the Great Barrier Reef, and land use changes since the 1850s have resulted in changes to the amount and the forms of nitrogen and sediment delivered to the Great Barrier Reef. For example, the predominant form of nitrogen was once dissolved organic nitrogen (DON), which originated from undisturbed landscapes (Harris, 2001; Brodie and Mitchell, 2005). Today, nitrogen is delivered mostly as dissolved inorganic nitrogen (DIN: nitrate + nitrite + ammonium) from fertiliser and sewage waste, and particulate nitrogen (PN) from soil erosion (Brodie and Mitchell 2005; Joo *et al.,* 2012; Turner *et al.,* 2012, in press; Kroon *et al.,* 2013). The shift to more bio-available forms of nitrogen has important consequences for Great Barrier Reef ecosystems (Fabricius 2005; Brodie *et al.,* 2012). Nitrate presents a higher risk as it is bio-available for in-stream uptake (e.g. weed growth) as well as for downstream uptake (e.g. promoting phytoplankton production and benthic algal growth on inshore reefs). The larger increases in the inorganic nitrogen fraction are associated with intensive fertiliser use on sugarcane and banana crops in these catchments (Armour *et al.,* 2009).

Most suspended solids and particulate nutrients settle quickly from the plume and are deposited within a few kilometres of the river mouth (e.g. for the Fitzroy River see Webster and Ford, 2010). In plumes from the Burdekin River, suspended solids concentrations can drop from more than 500 milligrams per litre in the river at zero salinity close to the river mouth to less than 10 milligrams per litre at salinities around 5 to 10 practical salinity units (Bainbridge *et al.,* 2012). Fine benthic sediment is continuously resuspended in shallow waters (less than 10 metres deep) by the prevailing south east wind regime and tidal currents and transported north along the coast (Larcombe *et al.,* 1995; Radke *et al.,* 2010). Coarser sediments are mostly retained near the coast in sand bars, beach ridges and sub-littoral dunes (Ryan *et al.,* 2007) often consolidated by mangrove recruitment). Recent work (described in Turner *et al.,* in press) has shown that different catchments have statistically different particle size distributions – with the Fitzroy, Plane, Burnett, Normanby and Burdekin catchments having a greater percentage of clays (less than four micrometres), while the Barron, Tully, Barratta, Johnstone, Herbert and Pioneer catchments have a larger percentage of silt (four to 62 micrometres), fine sand (62 to 250 micrometres) and coarse sand (250 to 2000 micrometres). This suggests that the first group of catchments are likely to contribute proportionally more fine particles to the Great Barrier Reef lagoon and that these are likely to affect a greater area of the Great Barrier Reef lagoon.

In contrast, dissolved nutrients move conservatively through the low salinity areas of the estuarine plume, indicating very little biological uptake because phytoplankton growth is light limited in the highly turbid plume waters (suspended solids more than 10 milligrams per litre) (Bainbridge *et al.,* 2012; Turner *et al.,* 1990). However, in areas of higher salinity (25 to 36 practical salinity units), which exist towards the offshore boundary of the plume, there is higher biological uptake of nutrients (Brodie *et al.,* 2010; Devlin and Brodie, 2005; Devlin and Schaffelke, 2009). Satellite images and plume sampling reveal that algal blooms develop when turbidity declines to values of less than 10 milligrams per litre and clear skies allow increased phytoplankton photosynthesis (usually two to five days after the peak discharge occurs) (Bainbridge *et al.,* 2012; Brodie *et al.,* 2010; Fabricius *et al.,* 2012). Thus, inorganic nutrients can be transported over long distances, often greater than 50 kilometres, exposing inshore reefs and seagrass meadows to high inorganic nutrient concentrations for short periods of time (Brodie *et al.,* 2006; Devlin and Brodie, 2005; Devlin *et al.,* 2010; Rohde *et al.,* 2006; Faithful *et al.,* 2008; Schroeder *et al.,* 2009).

Perhaps the most important factor determining susceptibility of coastal ecosystems to adverse effects from land-derived pollutants is the amount of exchange between the water body and the open ocean. Longer water residence times promote an increasing build-up of land-derived pollutants, thus water bodies with low exchange rates with the open ocean seem to be particularly vulnerable to the effects of pollution. There have been a number of previous studies of residence times or flushing times of water (and implicitly of pollutants) in the Great Barrier Reef lagoon (Hancock *et al.,* 2006; Luick *et al.,* 2007; Wang *et al.,*2007; Choukroun *et al.,*2010). The consensus of all these studies is that water residence times for the Great Barrier Reef are relatively short and rapid flushing with residence times of the order of weeks up to possibly one year. However, Andutta *et al.,* (2011) have shown that in bays along the coast, hypersaline waters escape sideways by the residual longshore southward currents and are transferred from bay to bay until steady-state conditions are reached after about 100 days, indicating a long residence time of inshore waters influencing the impact of the terrestrially derived pollutants.

Studies using remotely-sensed data have contributed to our understanding of the occurrence and influence of river plumes, and the ability to estimate exposure of marine ecosystems to land-based pollutants. The combination of traditional concurrent surface water sampling and image analysis of plume frequency distribution for parameters such as suspended sediments and chlorophyll *a* (Devlin and Schaffelke 2009) has allowed for the quantitative assessment of the degree of exposure of reefs (and other ecosystems) to flood waters (Brodie *et al.,* 2007, 2010; Wooldridge *et al.,* 2006; Devlin *et al.,* 2010, 2012; Rohde *et al.,* 2006; Alvarez-Romero *et al.,* 2013). In addition, there are now more sophisticated tools such as the SLIM oceanography model and the HOME eco-hydrology model (Costanza *et al.,* 2011; Lambrechts *et al.,* 2010; Richmond and Wolanski 2011), the ChloroSim model (Wooldridge *et al.,* 2006) and the SHOC (Sparse Hydrodynamic Ocean Code; http://www.emg.cmar.csiro.au/www/en/emg/software/EMS/hydrodynamics.html) model (Herzfeld *et al.,* 2006; Robson *et al.,* 2008) being applied to assess exposure and response of marine ecosystems to land run-off in the Great Barrier Reef. The SHOC model has recently been developed into a three-dimensional baroclinic hydrodynamic model of the whole Great Barrier Reef lagoon with accurate forcing for offshore ocean boundaries as part of the eReefs project (see http://www.bom.gov.au/environment/activities/coastal-info.shtml), based primarily on the hydrodynamic model.

Advances in mapping and modelling reiterate that the inner-shelf seagrasses and reefs adjacent to the high-rainfall Wet Tropics region are exposed regularly to a combination of low salinity water, suspended sediments, particulate and dissolved nutrients and pesticides (Bainbridge *et al.,* 2009; Devlin *et al.,* 2012b; Devlin and Schaffelke 2009; Lewis *et al.,* 2012b; Wolanski *et al.,* 2008). An analysis of freshwater extent in Great Barrier Reef waters from 2002-2010, utilising remotely sensed coloured dissolved organic matter (CDOM) as a proxy, indicates that both inner and mid-shelf reefs can be affected with a maximum area of influence of 22,034 square kilometres within this period (Schroeder *et al.,* 2012). Integration of ocean colour imagery combined with the frequency of plume occurrence and spatially distributed pollutant loads identify large areas of the inshore Great Barrier Reef being exposed to elevated surface dissolved inorganic nitrogen and total suspended solid concentration sources from river plumes (Alvarez-Romero *et al.,* in press). During severe floods in 2011, the maximum areas exposed to high concentrations of dissolved inorganic nitrogen and total suspended solids were 22,149 square kilometres and 5860 square kilometres, respectively (Alvarez-Romero *et al.,* in press). The larger surface area exposed to dissolved inorganic nitrogen reflects the offshore movement of dissolved nitrogen. There are substantive differences in exposure of different ecosystems to terrestrial pollutants due to the differences in the movements of pollutants and the location of sensitive ecosystems, such as coral reefs and seagrass meadows.

Spatial and temporal distribution of pollutants in the Great Barrier Reef

The water quality in the Great Barrier Reef lagoon varies greatly over time and space. There are distinct cross-shelf gradients from the coast to the shelf-break with mean concentrations of particulate water quality variables such as suspended solids (SS), particulate nutrient and carbon (particulate nitrogen, particulate phosphorus, particulate carbon) , chlorophyll *a* and Secchi depth (a measure of water clarity) being 0.6- to 5-fold higher in the coastal and inshore areas compared to the offshore lagoon, especially adjacent to the Wet Tropics and Burdekin regions in the central Great Barrier Reef (Brodie *et al.,* 2007; De'ath and Fabricius, 2008, 2010; McKinnon *et al.,* in press). These gradients are less pronounced in the Far Northern region (Brodie *et al.,* 2007; De'ath and Fabricius 2008, 2010) and also less clear for dissolved nutrients which are short-lived in the lagoon due to rapid biological uptake and transformation (Furnas *et al.,* 1995, 1997, 2005). Water quality gradients away from the coast, albeit less pronounced, are also evident in the coastal and inshore lagoon (Cooper *et al.,* 2007; Schaffelke *et al.,* a, b 2012). Mid and outer shelf concentrations of chlorophyll *a* generally increase towards the south, but not along a consistent gradient, and are likely to be driven by factors such as upwelling, wind and tidal currents (Brodie *et al.,* 2007). However, a model of the catchment-to-lagoon connection has shown a strong correlation between the latitudinal gradient of chlorophyll *a* and the concentration of dissolved inorganic nitrogen in riverine flood waters (Wooldridge *et al.,* 2006).

Seasonality is the strongest determinant of water quality in the Great Barrier Reef lagoon. During winter, concentrations of water quality variables, especially nutrients, are very low (i.e. close to or below detection limits), except for periods of strong winds leading to resuspension of settled fine sediments which intermittently increases water turbidity and releases nutrients (Schaffelke *et al.,* 2012a, b,). Compared to other coral reef areas (e.g., the Caribbean and Florida), mean water column concentrations of suspended solids, chlorophyll *a* and soluble reactive phosphorus are higher in the inshore Great Barrier Reef, while concentrations of dissolved nitrogen are lower (see references in Schaffelke *et al.,* 2012a).

During summer, most of the routinely measured water quality variables can change dramatically for several weeks in response to disturbance events such as river floods, cyclones and, especially affecting the outer lagoon, shelf-break upwelling (Schaffelke *et al.,* 2012a; McKinnon *et al.,* in press). Much elevated concentrations of nutrients (both dissolved and particulate) and suspended solids are observed in coastal and inshore waters after exposure to riverine flood plumes (Devlin and Brodie 2005; Devlin and Schaffelke 2009; Brodie *et al.,* 2010; Devlin *et al.,* 2012; Schaffelke *et al.,* 2012a) and after re-suspension of bottom sediments by strong winds or tidal currents (Furnas, 1989, Schaffelke *et al.,* 2012 a,b). In recent years (2008-2011), above-median river flow led to significant, in some regions record-breaking, floods (Turner *et al.,* in press) and several severe cyclones affected the Great Barrier Reef and coast. These events have left a clear signal in the inshore water quality with most variables showing elevated values, except salinity which decreases (Devlin *et al.,* 2012; Schaffelke *et al.,* 2012 b). Water quality monitoring in future dry years will show how long the flood signal is discernible.

Although regular monitoring of the inshore water quality has been carried out since 2005 (Reef Rescue Marine Monitoring Program) there are limited water quality data available for the Great Barrier Reef to detect long-term, i.e. decadal, trends. A recent analysis of the Cairns Long-term Water Quality Transect data, sampled by the Australian Institute of Marine Science (AIMS) since 1989, showed clear relationships between concentrations of marine water quality variables and several environmental factors, including land use on the adjacent Barron, Daintree and Russell/Mulgrave river catchments (Schaffelke *et al.,* 2010). Most variation in water quality was explained by seasonal differences. However, high catchment vegetation clearing rates from 1996-2001, together with high river discharge were associated with high concentrations of chlorophyll *a*, suspended solids, particulate nitrogen and particulate phosphorus in the adjacent coastal marine waters. After 2004, much less land clearing occurred and concentrations of particulate water quality variables remained constant or decreased. This is the first time that the direct correlation between land-use change on the catchment (land clearing rate) and marine water quality has been measured, underlining the importance of long-term data series.

The inorganic nutrients imported or released during disturbance events are rapidly taken up by pelagic and benthic algae and microbial communities (Alongi and McKinnon 2005), often leading to high levels of organic production and short-lived phytoplankton blooms (Furnas 1989; Furnas *et al.,* 2005, 2011). Via these processes, nutrients from land runoff augment overall regional stocks of nutrients already stored in sediments, biomass or detritus (Furnas et al 2011). However, most of the inorganic nutrients used by marine plants and bacteria on a day-to-day basis come from recycling of nutrients already within the Great Barrier Reef ecosystem (Furnas *et al.,* 2005; Furnas *et al.,* 2011). The produced organic matter is cycled through the marine food web and transformed into detrital particles (e.g. marine snow) that may be deposited on benthic communities, such as coral reefs, and influence their structure, productivity, and health for long periods (reviewed in Fabricius 2011). Marine snow particles are more abundant closer to the coast, especially during summer (Uthicke *et al.,* 2009). By cycling through the food web, event-driven inputs of nutrients are likely to not only have an acute but also a longer-term ecosystem effect. However, organic matter accumulation is complex, dependent on both input and transformation processes as well as hydrodynamics. We can currently only speculate how long the influence of a flood event lasts and how it is perpetuated through the food web. While wind-driven resuspension is one of the major drivers of turbidity in the inshore Great Barrier Reef lagoon (e.g. Larcombe *et al.,* 1995; Wolanski 2007, Orpin and Ridd, 2012), recent research indicates that flood-delivered fine sediment remains in the coastal zone long after the event, leading to recurring high turbidity events through wind-driven re-suspension (Wolanski *et al.,* 2008; Lambrechts *et al.,* 2010; Fabricius *et al.,* 2013a).

Potential changes to the trophic status of the Great Barrier Reef are not yet fully understood. A detailed pelagic metabolism study confirmed earlier work that that the Great Barrier Reef lagoon in general is moderately productive, but can show high production during events such as flood intrusions, cyclonic mixing or upwelling (McKinnon *et al.,* in press). Great Barrier Reef lagoon areas affected by such events could be considered episodically eutrophic (Brodie *et al.,* 2011; McKinnon *et al.,* in press). Persistent elevated chlorophyll values were only found at some routine water quality monitoring sites (Schaffelke *et al.,* 2012 a,b). However, results from remote sensing indicate that high mean and median chlorophyll values (relative to the Great Barrier Reef Water Quality Guidelines; GBRMPA, 2009) occur widely throughout the inshore waters, especially in the Wet Tropics and Burdekin regions (Brando *et al.,* 2011). Without investigation of the associated pelagic productivity in these areas, it cannot be concluded that this is an indication of a widespread and chronically altered trophic status.

Waters of the Great Barrier Reef lagoon are polluted with a range of pesticides including herbicides, insecticides and fungicides. Pesticides, unlike nutrients, sediments and metals, have no natural sources. Their concentrations have been positively correlated with low salinity associated with river runoff and they appear in the marine environment in typical combinations of compounds that reflect the use on the adjacent catchments (Lewis *et al.,* 2009; Kennedy *et al.,* 2012a). Therefore, the occurrence of pesticides in the Great Barrier Reef can be attributed with great confidence to agricultural activities on the adjacent catchments. Modelled estimates of the total load of photosystem II inhibiting (PSII) herbicides range include 15,700 ± 830 kilograms per year (Lewis *et al.,* 2011), 16,700 kilograms per year (Waters *et al.,* in press) and 30,000 kilograms per year reported in Kroon *et al.,* (2012). Actual measured photosystem II inhibiting herbicide loads were four tonnes (for 2009-2010) and 12.5 tonnes (2010-2011), respectively (Smith *et al.,* 2012, Turner *et al.,* in press). The measured loads are smaller than the modelled loads as not all catchments are monitored. It should also be noted that these estimates underestimate the total load of pesticides being discharged to the Great Barrier Reef, as another 28 pesticides have been detected in Great Barrier Reef catchments (Smith *et al.,* 2012; Turner *et al.,* in press; Kroon *et al.,* 2013). Knowledge of the spatial and temporal extent of pesticide distribution in Great Barrier Reef waters has expanded considerably in recent years. Monitoring using the time-integrating passive sampler devices (Shaw and Müller 2005) found detectable levels of herbicides at all of the sampling sites in the inshore Great Barrier Reef lagoon and in estuarine waters throughout the year, generally with higher concentrations in the wet season (Kennedy *et al.,* 2012a,b; Shaw *et al.,* 2010). Herbicides generally also showed distinct cross-shelf gradients with higher concentrations close to the coast, as did other water quality constituents (see above). While herbicide concentrations do not exceed water quality guidelines during most of the year, they are still detectable, and the consequences for marine life of low level chronic exposure to a mixture of herbicides are unknown (see risk assessment in Lewis *et al.,* 2012a).

The detection of herbicides during the dry season is surprising. Only in the Wet Tropics region, where rivers are flowing throughout the year, albeit with much reduced flow during the dry season, can some export of herbicides be expected. At present, the persistence of pesticides in tropical marine water is very poorly understood, however, is the subject of current research under the National Environment Research Program (A. Negri, AIMS). It is possible that compounds adsorb to settling particulate matter, which would reduce light-dependent degradation, and are intermittently resuspended back into the water column by wind, tides and currents. Longer than expected persistence of herbicides is also indicated by finding detectable concentrations of herbicides in the Far Northern Great Barrier Reef (Kennedy *et al.,* 2012a), distant from their likely agricultural sources. These herbicides are likely transported northward by longshore water currents (e.g. Luick *et al.,* 2007) from agricultural catchment sources adjacent to the central Great Barrier Reef. Concentrations of pesticides during the dry season in the freshwater and estuarine reaches of Barratta Creek (part of the Bowling Green Bay Ramsar site), which receives irrigation tail-water from sugarcane and other cropping throughout the year, are consistently above guidelines and pose a considerable risk to the estuarine ecosystems of the Ramsar wetland (Davis *et al.,* 2012, in press; Smith *et al.,* 2012; Turner *et al.,* 2012, in press).

Changes in disturbance regimes

The Great Barrier Reef today is affected by more frequent and more intense acute disturbances compared to pre-industrial times. Mass coral bleaching is directly attributable to anomalously high seawater temperature (Hoegh-Guldberg *et al.,* 2007) driven to a large extent by rising concentrations of atmospheric greenhouse gases (Moss *et al.,* 2010). The frequency and severity of coral bleaching events are expected to increase over the course of this century (Donner 2009). Warming oceans are also resulting in increased intensity of tropical cyclones, although greater increases are forecast for the northern than southern hemisphere (Knutson *et al.,* 2010).

Global warming is also increasing rainfall variability (Lough, 2011) and this can affect runoff into the Great Barrier Reef lagoon. Increased sediment and nutrient losses from farms are particularly large when floods follow severe droughts as a result of low vegetation cover (McCulloch *et al.,* 2003; Packett *et al.,* 2009). Rivers draining into the central and southern rivers now carry five to nine-fold higher nutrient and sediment loads from cleared, fertilised, and urbanised catchments into the Great Barrier Reef compared to pre-European times (Kroon *et al.,* 2012). This increase in terrestrial runoff leads to substantial changes to coastal and inshore ecosystems (see below). Prolonged periods of low salinity resulting from increased freshwater runoff are likely to have implications for corals, which have limited tolerance to low salinity waters (Berkelmans *et al.,* 2012).

The already experienced increase in frequency of severe disturbance to Great Barrier Reef ecosystems means that recovery between disturbances is often incomplete leading to observed declines in ecosystem health and resilience (see Chapter 3 for a more in-depth discussion about the importance of extreme climatic events).

Condition of Great Barrier Reef ecosystems and response to land runoff

Land runoff from developed catchments influences the water quality in the coastal and inshore Great Barrier Reef. Increased riverine loads of nutrients, sediments and pesticides (Kroon *et al.,* 2012) have led to chronic changes in environmental conditions for Great Barrier Reef species and ecosystems since European settlement, as well as intermittent, acute changes after extreme river floods.

Understanding the effects of water quality changes on Great Barrier Reef species and ecosystems has improved considerably over the last decade due to a focused research effort and the ongoing monitoring of the condition of inshore coral reef and coastal seagrass ecosystems under the Marine Monitoring Program since 2005. Recent research has strengthened the evidence for causal relationships between water quality change and ecosystem health for corals, seagrasses and mangroves (reviewed in Brodie *et al.,*2012) and the effects of nutrient inputs and increased sedimentation on coral reefs and factors influencing the susceptibility and resilience of reefs to these stressors (reviewed in Fabricius 2011). Key outcomes of recent research, relevant to this chapter, are summarised below.

Coral reef ecosystems

Coral cover in parts of the Great Barrier Reef has been in decline for several decades (Bruno *et al.,* 2007; Hughes *et al.,* 2011; Sweatman *et al.,* 2011; Sweatman and Syms 2011). Large-scale and chronic disturbances have reduced coral cover on reefs in the central and southern Great Barrier Reef to less than 50 per cent of what it was in 1985, while the northern Great Barrier Reef has no apparent decline (De'ath *et al.,* 2012). While there are less data available, monitoring data from inshore reefs adjacent to the developed central and southern coast indicate that cover has declined on average by 34 per cent since 2005 (Thompson *et al.,* 2012). The causes of coral loss vary from reef to reef, depending on exposure to tropical cyclones, outbreaks of the crown-of-thorns starfish or coral disease, temperature anomalies causing coral bleaching, and exposure to floods (Osborne *et al.,* 2011; De'ath *et al.,* 2012; Thompson *et al.,* 2012). Observed long-term declines in coral cover indicate that the frequency and severity of disturbances is unsustainably high, with very few reefs avoiding severe loss of cover associated with at least one disturbance and most impacted by multiple disturbances (De'ath *et al.,* 2012; Thompson *et al.,* 2012). Against this background of disturbances, the persistence of coral communities is reliant on high resilience, i.e. on maximising both resistance to disturbances and ability to recover during intervening periods (Thompson and Dolman 2010; De'ath *et al.,* 2012). Poor water quality as a result of sediments, nutrients and pesticides delivered to the inshore reefs in runoff has the potential to both increase the susceptibility of corals to disturbances and then suppress their subsequent recovery.

Water quality and coral reef community composition

Turbidity, nutrient availability and sedimentation shape the composition of benthic communities on coastal and inshore reefs of the Great Barrier Reef (Done 1982; van Woesik and Done 1997; van Woesik *et al.,* 1999; Fabricius *et al.,* 2005; De'ath and Fabricius 2008; Browne *et al.,* 2010; Death and Fabricius 2010; Thompson *et al.,* 2010; Uthicke *et al.,* 2010; Browne *et al.,* 2012; Fabricius *et al.,* 2012). Along the Queensland coast, differences in turbidity and sedimentation patterns largely result from the resuspension of accumulated seabed sediment deposits by waves and tidal currents (Larcombe *et al.,* 1995; Larcombe *et al.,* 2001; Wolanski *et al.,* 2005; Orpin and Ridd 2012). Superimposed on these ambient conditions are the additional fluxes of sediments, nutrients and pesticides delivered in catchment runoff (Fabricius *et al.,* 2013a; Lewis *et al.,* 2012a; Kroon *et al.,* 2012). Experimental research has demonstrated how water quality parameters, for example chronic herbicide exposure, can influence species composition in tropical marine communities (Magnusson *et al.,* 2012). However, not all reefs are equally vulnerable to degradation from exposure to land run-off and pollution; the combination of exposure to nutrients, sediments and pollutants, site-specific environmental conditions and species composition determine their susceptibility. Based on international case studies, the most vulnerable reefs are close to river mouths (especially adjacent to disturbed land areas and deeper parts of reefs); poorly flushed reefs; reefs on shallow continental shelf areas (especially if surrounded by fine, soft sediment areas prone to resuspension); reefs frequently disturbed in the past (e.g. by severe storms and coral bleaching) and reefs with low abundances of herbivorous fish (Fabricius 2011). The effects of the various water quality parameters are discussed in detail below.

Light – turbidity - sedimentation

Most reef building corals gain energy via the photosynthetic products of symbiotic algae and, as such, are reliant on light (e.g. Muscatine 1990). While corals can occur across a broad range of light conditions due to their physiological (e.g. Falkowski *et al.,* 1990; Titlyanov 1991; Anthony and Fabricius 2000) and morphological (Anthony *et al.,* 2005; Todd 2008) plasticity, some corals are vulnerable to reduced light levels, leading to critical metabolic deficits (Cooper *et al.,* 2008).

Reduced light penetration in the water column is directly proportional to the concentration and provenance of suspended particles (sediments and plankton; Te 1997) and coloured dissolved organic matter, which can vary by up to two orders of magnitude within days to weeks with dramatic consequences for benthic light conditions (Anthony *et al.,* 2004). The influence of river floods on turbidity close to inshore reefs increases with increasing proximity to a river mouth (Fabricius *et al.,* 2013a). Also, for a given wave height and tidal flow, Great Barrier Reef inshore turbidity is 13 per cent higher in weeks with high compared to low river flow and rainfall, and there is a 28 per cent decline in turbidity across the dry season (Fabricius *et al.,* 2013a). This, along with the demonstrated transport of fine sediments in flocculated form from river mouths far into inshore waters (Bainbridge *et al.,* 2012), clearly demonstrates that runoff can chronically and episodically reduce the levels of light reaching corals on coastal reefs.

Rates of sediment deposition on corals and surrounding substrata depend on the supply of suspended particles and the local hydrodynamics, which jointly determine the balance between particle settlement and resuspension (Wolanski *et al.,* 2005; Browne *et al.,* 2012). Corals vary greatly in their tolerance to sediment deposition, with some of this variation due to differences in colony morphologies (review by Erftemeijer *et al.,* 2012; Flores *et al.,* 2012). Tolerance can also be considered in terms of the energetic cost to the colony. The energy required to shed sediments varies between species due to differences in the efficiencies of passive (largely dependent on growth form) or active (such as cilia or tentacle manipulation or mucus production) strategies for sediment removal (Rogers 1990; Stafford-Smith and Ormond 1992; Sofonia and Anthony 2008). Compounding the energy cost of removal is the reduction of photosynthesis by sediments that shade the coral (Weber *et al.,* 2012).

The nature of sediments also influences effects on corals, with nutrient-enriched fine-grained terrestrial silts being particularly detrimental as they are more difficult for corals to shed, have relatively large surface area facilitating the adsorption of nutrients and pesticides, and, when settled onto coral tissue, cause a barrier to gas exchange and rapidly alter microbial communities leading to tissue mortality (Weber *et al.,* 2006; Weber *et al.,* 2012). In the Burdekin River plume, fine grained sediments are entrained by planktonic and transparent exopolymer particulate matter and transported as large sticky aggregates (Bainbridge *et al.,* 2012); a form of sediment most likely to cause harm to corals. Indeed, observations of increased levels of coral disease following periods of flooding at both individual reefs (Haapkylä *et al.,* 2011) and at regional scales (Thompson *et al.,* 2011) strongly implicate runoff as a factor increasing corals' susceptibility to disease.

Early life-history stages of corals have been repeatedly shown to be particularly vulnerable to sedimentation (reviewed by Fabricius 2005). Sediment accumulation reduces coral settlement by physically blocking access to suitable substrate (Birrell *et al.,* 2005) but also altering benthic microbial communities and so disrupting critical chemical cues for settlement and metamorphosis

(Negri *et al.,* 2002; Webster *et al.,* 2004). Once settled, juveniles are highly vulnerable to sedimentation due to their small size, which precludes passive shedding and small energy reserves limiting the scope for active removal or tolerance of intermittent shading (Fabricius and Wolanski 2000). The results of increased sedimentation have been observed on Whitsunday Island reefs, where, following a period of high river flows and associated increase in sediment and nutrient loads (Joo *et al.,* 2012), the densities of juvenile corals on reefs prone to sediment accumulation have declined markedly (Thompson *et al.,* 2012).

A number of studies have examined the growth dynamics of coastal turbid reefs, either using the history of the reefs through the Holocene (from reef cores) or short term (a few years) studies on the reefs themselves to assess their response to current turbid conditions and how this may have changed with the anthropogenic effects of increased sediment delivery to coastal waters. The reefs include Dunk Island near Tully (Perry and Smithers; 2010; Perry *et al.,* 2011); Middle Reef near Townsville (Browne *et al.,* 2010; Browne 2012; Perry *et al.,* 2012); Paluma Shoals north of Townsville (Perry and Smithers 2006; Perry *et al.,* 2008; Palmer *et al.,* 2010); King Reef near Tully (Roche *et al.,* 2011); Lugger Shoals near Tully (Perry and Smithers, 2006; Perry *et al.,* 2009); Nelly Bay reef, Magnetic Island (Lewis *et al.,* 2012b) and several of these reefs in combined analyses (Perry and Smithers, 2011; Browne *et al.,* 2012).

Overall it appears most of these coastal reefs have communities of taxa tolerant to the highly turbid conditions. Over the Holocene these reefs have experienced periods of growth activity and stagnation, depending on environmental conditions (see also Kleypas 1996). Due to the high turbidity tolerance, these reefs may not be as affected by the increased loads of terrestrial sediment from anthropogenic catchment erosion. Corals in very shallow water (two to three metres) still receive light in sufficient quantities for photosynthesis at low tide even in very turbid water and are also able to use heterotrophic particle feeding to supplement their food requirements (Anthony and Fabricius 2000). Terrestrial sediment is also incorporated into the reef framework on these reefs in addition to coral-derived carbonate (Perry *et al.,* 2012) with silicoclastic material comprising 20 to 50 per cent of the reef material at Nelly Bay (Lewis *et al.,* 2012b). The growth of coral species that tolerate high turbidity and sedimentation facilitates rapid vertical reef framework accumulation on these reefs driven by both high rates of sedimentation infill and carbonate production. This likely only occurs in locations where the coral communities are already adapted to deal with high sedimentation and to the periodically low light conditions associated with fluctuating turbidity and tidal daytime exposure (Browne *et al.,* 2012; Perry *et al.,* 2012).

In contrast, recent studies indicate that reefs further off the coast have undergone dramatic changes in coral species composition and abundance associated with multiple recent disturbances including increased terrestrial runoff (Roff *et al.,* 2013). Dating of dead *Acropora* fragments indicated a previously undocumented historical collapse of *Acropora* assemblages at Pelorus Island (central Great Barrier Reef) occurring between 1920 and 1955, with few observations of *Acropora* fragments in the samples at dates later than 1980 (Roff *et al.,* 2013). Further, the study indicated that the inshore reef exhibited long-term stability in coral community structure over centennial scales prior to 1920. Roff *et al.,* (2013) suggest that increases in sediment and nutrient loading following European settlement acted as the ultimate cause for the lack of recovery of *Acropora* assemblages following a series of acute disturbance events (high water temperatures, cyclones and flood events). Similar to coral communities, benthic foraminifera communities in sediment cores from inshore reefs of the Great Barrier Reef showed high persistence over several thousand years (Uthicke *et al.,* 2012a). However, foram communities on reefs close to the source of runoff exhibited significant changes after European settlement of North Queensland, whereas communities further from the coastline remained stable, which provides further evidence that land use changes have led to significant changes in water quality which in turn affects marine communities.

Nutrients

High availability of dissolved inorganic nutrients can lead to significant physiological changes such as decreased calcification and higher concentrations of photopigments, but generally does not directly lead to hard coral mortality (reviewed in Fabricius 2005, 2011). Dissolved inorganic nutrients are rapidly taken up by bacterio- and phytoplankton and benthic algae before they would reach coral in high enough concentrations to affect corals directly. However, the indirect effects of nutrient enrichment on corals are now better understood, for example enhancing the productivity and growth of competing macroalgae and phytoplankton which in turn leads to high organic matter concentrations in the water column and in sediments.

For corals on the Great Barrier Reef as a whole the most pressing threat of nutrients is the link between nutrient supply and outbreaks of a native pest. Increased phytoplankton productivity and biomass in the lagoonal water column following floods increases the likelihood of outbreaks of the crown-of-thorns starfish in the central Great Barrier Reef as it enhances their larval survival (Fabricius *et al.,* 2010). Statistical models suggest that crown-of-thorns starfish outbreaks have increased from an historical frequency of once in 50 to 80 years to the recent pattern of once in every approximately 15 years; observed since the 1960s (Brodie *et al.,* 2005; Fabricius *et al.,* 2010).

Nutrient enrichment can promote growth of macroalgae where light levels are sufficient (Schaffelke 1999) or of heterotrophic filter feeders (Johannes *et al.,* 1983; Birkeland, 1988). Macroalgae have higher abundance in areas with high water column concentrations of chlorophyll, indicating high nutrient availability from runoff (De'ath and Fabricius 2010). High macroalgal abundance has been shown to suppress reef resilience (e.g., Hughes *et al.,* 2007, Cheal *et al.,* 2010; Foster *et al.,* 2008), with the influence of macroalgae being inversely proportional to the size of the corals they compete with (Ferrari *et al.,* 2012). In addition to simple competition for space and shading of corals (e.g. McCook *et al.,* 2001) macroalgae alter water chemistry and microbial communities, affecting coral metabolism (Hauri *et al.,* 2010), settlement cues for coral larvae (Birrell *et al.,* 2008), and susceptibility to disease (Morrow *et al.,* 2012). The frequency of coral diseases also appears to be linked to warming (Bruno *et al.,* 2007) and elevated concentration of nutrients (Bruno *et al.,* 2003).

Because elevated turbidity and nutrient availability are usually correlated during runoff events it is often difficult to ascertain the importance of the individual stressors. Studies on foraminifera distribution on inshore areas of the Great Barrier Reef concluded that elevated nutrients are the limiting factor for the distribution of many symbiont-bearing species (Nobes *et al.,* 2008; Uthicke and Nobes 2008) and experiments showed that growth of symbiont-bearing foraminifera is reduced in inshore reefs subjected to high runoff (Uthicke and Altenrath 2010; Reymond *et al.,* 2011). Because of the clear community response of foraminifera to water quality decrease and other pollution (reviewed in Reymond *et al.,* 2012) an index based on their community composition (FORAM index) is used as additional tool to assess reef health in the Marine Monitoring Program (Uthicke and Nobes 2008; Uthicke *et al.,* 2010; Thompson *et al.,* 2012).

There are several lines of evidence based on physiological or correlative field data suggesting that corals in nutrient-enriched or turbid waters are more vulnerable to temperature stress than those in low nutrient environments:

- Corals exposed to high levels of nutrients have elevated bleaching susceptibility compared with corals in low nutrient environments. High concentrations of nitrate lead to high endosymbiont (zooxanthellae) division rates, resulting in a relative under-supply of phosphate, which leads to altered cell structures with enhanced susceptibility to thermal stress and light damage (Wiedenmann *et al.,* 2012).

- High nutrients can reduce the photosynthetic efficiency and light-harvesting capacity of algal endosymbionts (Dubinsky *et al.,* 1990), possibly exacerbating heat damage to them (Nordemar *et al.,* 2003).
- Elevated ratios of endosymbiont to host cell densities in nutrient-exposed corals have been associated with increased bleaching susceptibility, possibly because the production of harmful reactive oxygen species increase with symbiont densities (Cunning and Baker 2012).
- Two common species of corals on the inshore Great Barrier Reef showed significantly greater photophysiological stress and lower survival in coastal water that was organically enriched with nutrients and resuspended sediments than those in coastal water that was not organically enriched Fabricius *et al.,* (2013b).
- Spatial correlations between nutrient status and bleaching history on the Great Barrier Reef suggest that inshore reefs had lower bleaching resistance per degree heating week than offshore reefs during the 1998 and 2002 bleaching events (Wooldridge 2009; Wooldridge and Done 2009). The proposed mechanism is centred on dissolved inorganic nitrogen disrupting the trophic balance between algal endosymbionts and coral hosts at high temperatures and light. Temperature tolerance of corals at elevated dissolved inorganic nitrogen concentrations is suggested to reduce by two to 2.5 degrees Celsius, and that a 50 to 80 per cent reduction in river nutrient discharges would ‘buy time’ for the reef in a progressively warming climate (Wooldridge *et al.,* 2006).
- Corals may experience up to 0.6 degrees Celsius greater colony surface temperatures in high nutrients, due to increased photo-pigment densities (darker surfaces absorb more light energy; Dubinsky *et al.,* 1990, Fabricius 2006, Jimenez *et al.,* 2008). Corals in turbid water rarely experience high light stress (Goreau *et al.,* 2000; West and Salm 2003). However, during bleaching conditions waters can be unusually calm and clear and the generally low light-acclimatized and darkly pigmented inshore colonies are likely to experience greater photoinhibition than light-acclimatized and paler offshore colonies.
- Calcification in massive *Montastraea faveolata* is reduced for longer after heat stress in turbid inshore compared to cleaner offshore reefs at similar levels of heat exposure (more than eight versus two to three years)(Carilli *et al.,* 2009; Carilli, 2010).
- Finally, some corals on turbid inshore reefs switch their symbiotic partners to clade D algal endosymbionts (van Oppen *et al.,* 2009), which provides a more heat stable symbiosis resisting temperatures approximately one degree Celsius higher than those tolerated by corals hosting clade C endosymbionts. Although less vulnerable to bleaching, corals hosting clade D symbionts have chronically reduced growth rates compared with conspecifics hosting clade C symbionts; hence these more heat tolerant colonies are competitively disadvantaged in normal temperature regimes.
- Foraminifera with the same symbionts as corals (*Marginopora vertebralis*) showed lowest growth rates and highest mortality under elevated temperature and increased nitrate concentrations (Uthicke *et al.,* 2012b). In contrast bleaching thresholds of those with diatoms as symbionts were not affected by increased dissolved inorganic nitrogen (Schmidt *et al.,* 2011).

In contrast, other studies suggest that bleaching risk during temperature and high light stress can be ameliorated by nutrient supply. These studies tended to focus on relatively oligotrophic environments and/or used experimental starvation of corals, conditions that cannot be extrapolated to predict bleaching outcomes on more eutrophic Great Barrier Reef inshore reefs. They showed that corals are more resistant (later onset of bleaching, longer tolerance of high temperatures before bleaching occurs, and milder symptoms) and resilient (having higher rates of survival and recovery from bleaching) when they are well-fed compared to those that were experimentally starved (Borell *et al.,* 2008; Ferrier-Pages *et al.,* 2010; Hoogenboom *et al.,* 2012; Borell and Bischof 2008; Fitt *et al.,* 2000; Grottoli *et al.,* 2004). Also, species capable of up-regulating heterotrophy when bleached recover faster from bleaching than predominantly phototrophic species (Grottoli *et al.,* 2006; Anthony *et al.,* 2009). Importantly, colonies of *Acropora intermedia* had lower bleaching and higher survival rates when supplied with suspended particulate matter compared to those not exposed to this source of food and shading (Anthony *et al.,* 2007), or when supplied with rotifers compared to colonies in filtered water (Connolly *et al.,* 2012).

Fabricius *et al.,* (2013b) propose a conceptual trophic framework that synthesizes the apparently contradictory outcomes of the different studies that suggest either greater or reduced thermal tolerance in response to changes in nutrient status. The framework illustrates two important points: First, nutrients and light can represent either a stress or a beneficial factor (Hoogenboom *et al.,* 2009), with optimum responses at intermediate rates and detrimental effects at too high and too low exposures (with absolute values being species-specific). Second, shifts in the trophic status of the environment (from oligotrophic to eutrophic) do not easily translate into shifts in the trophic status of reef corals (from starved to well-fed), because the types of food utilized and trophic plasticity vary greatly between species (Anthony and Fabricius 2000; Grottoli *et al.,* 2006; Houlbrèque and Ferrier-Pages 2009). Fabricius *et al.,* (2013b) conclude that in more eutrophic environments, as found on parts of the inshore Great Barrier Reef south of Cooktown, the exposure to additional nutrients is predominantly a stress factor for most coral species. The review re-confirms that improvement of water quality enhances the resilience of Great Barrier Reef coral reefs through several means, including improved tolerance to thermal stress.

Pesticides

Several persistent and mobile photosystem II inhibiting herbicides are regularly detected in the Great Barrier Reef lagoon. During flood events, photosystem II inhibiting herbicides have been detected at concentrations exceeding regulatory guidelines and at concentrations known to affect microalgae, corals and seagrass (Lewis *et al.,* 2009; Lewis *et al.,* 2012a; Smith *et al.,* 2012) and therefore represent a threat to these inshore primary producers. Photosystem II inhibiting herbicides reduce the efficiency of photosynthesis and in the longer term can cause damage to photosynthesis processes in corals (Jones and Kerswell, 2003; Negri *et al.,* 2005), microalgae (Bengtson Nash *et al.,* 2005; Magnusson *et al.,* 2008), crustose coralline algae (Negri *et al.,* 2011); foraminifera (van Dam *et al.,* 2012b) and seagrass (Haynes *et al.,* 2000; Gao *et al.,* 2011). The reduced photosynthesis in algae due to herbicide exposure causes reductions in the growth of these algae (Magnusson *et al.,* 2008) and changes in species composition (Magnusson *et al.,* 2012) but the effects of chronic exposures in inshore environments remain largely unknown.

Photosystem II inhibiting herbicides reduce energy acquisition by the host coral, which can lead to reduced reproductive output (Cantin *et al.,* 2007; 2009). Comparison to water quality guidelines gives an indication of the magnitude of adverse effects for individual chemicals. However, the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC and ARMCANZ, 2000) recommend that when multiple chemicals occur simultaneously in a waterbody their combined toxicity should be estimated using the concentration addition model. This model states, that any concentration of a contaminant (even if a biological effect cannot be detected for that contaminant acting alone at that concentration) contributes to the toxicity of the mixture (Kortenkamp *et al.,* 2009). This is sometimes referred to as the “something from nothing” phenomena. There is considerable experimental support of this hypothesis (Könemann 1980; Hermens *et al.,* 1984, 1985a,b; Broderius and Kahl 1985) including work on the toxicity of PSII herbicides on reef micro-algae (Shaw *et al.,* 2009; Magnusson *et al.,* 2010) and for using this as the default model for estimating mixture toxicity (e.g. Warne and Hawker, 1995; Kortenkamp *et al.,* 2009). In addition to the toxicity of mixtures of pesticides, their interaction with other contaminants, including emerging chemicals (see later section), total suspended solids, nutrients, and/or physico-chemical properties (e.g. light intensity and temperature) have largely been ignored despite there being some limited information supporting such effects. Harrington *et al.,* (2005), for example, found that not only did the combined effects of diuron with fine sediments amplify the inhibition of photosynthesis in crustose coralline algae, but the combined exposure retarded the ability of the crustose coralline algae to recover (compared to the effect of the stressors on their own).

The cumulative effects of repeated exposure of coral ecosystems to pesticides, as well as other contaminants, is not known, but would be expected to be greater than single exposures. If each exposure exerts a deleterious effect, and if the period between exposures is shorter than the period needed for full recovery, then the health of the ecosystem will gradually decline consistent with the predictions of the threshold damage model and damage assessment model. While these models were developed to explain the results on individual populations, their concepts should be broadly applicable to ecosystems. The threshold damage model has been used to predict the effects from fluctuating and pulsed exposures of organisms to single contaminants and mixtures (Ashauer *et al.,* 2006, 2007a, 2007b; Ashauer *et al.,* 2010b); while the damage assessment model has been used to model the effects of mixtures of internal metabolites (Lee and Landrum 2006a; 2006b).

There are multiple high volume events each year in most catchments except in dry *El Niño* periods (Turner *et al.,* in press) and as the distance from the mouth of rivers increases the period between exposure to elevated pesticides will increase and decrease the potential cumulative impact. At this stage it is not possible to quantify the potential cumulative effects but a first attempt to do this is presented in Lewis *et al.,* (2013). Another limitation to work done to date is that it has focused on photosystem II inhibiting herbicides but many other pesticides are present and are likely to contribute to detrimental effects on Great Barrier Reef ecosystems. These above factors of mixtures and the cumulative impacts of repeated exposures apply equally to the potential impacts of pesticides on seagrass beads and mangroves (see below).

The highest herbicide concentrations usually occur during the warmest months on the Great Barrier Reef and recent studies have demonstrated that photosystem II inhibiting herbicides exposure significantly increases the sensitivity of corals and foraminifera to thermal stress (Negri *et al.,* 2011; van Dam *et al.,* 2012a) and this may also contribute to coral bleaching.

Corals and fish are also very sensitive to insecticides applied within Great Barrier Reef catchments (Markey *et al.,* 2007; Botté *et al.,* 2012); however, the higher loads, greater mobility and potentially longer persistence of photosystem II inhibiting herbicides mean that these contaminants are likely to pose a greater immediate threat to inshore Great Barrier Reef ecosystems. Despite this, the persistence of these herbicides and their chronic effects on key inshore species such as seagrasses remain largely unknown.

The large number of insecticides and herbicides registered for use in Great Barrier Reef catchments that are currently not monitored and have not been tested for toxicity adds further uncertainty to the impact of pesticides on Great Barrier Reef ecosystems (King *et al.,* 2012). While it seems unlikely that pesticides alone pose a major threat to the entire Great Barrier Reef, we conclude that efforts to significantly reduce pesticide contamination of the Great Barrier Reef should continue based on the connectivity of catchment, estuarine and Great Barrier Reef ecosystems, the high sensitivity of foundation species to pesticides coupled with the widespread contamination of Great Barrier Reef waters and sediments, and emerging evidence that pesticides increase the vulnerability of Great Barrier Reef species to the negative effects of climate change.

Other environmental drivers

Ocean acidification is becoming increasingly recognized as an additional pressure on the Great Barrier Reef and on coral reefs globally. Understanding how coastal influences affect the pH and carbonate chemistry of inshore, mid shelf and offshore reefs, and processes interacting with ocean acidification, will be critical for the management of the Great Barrier Reef in the future. On offshore reefs, the seawater chemistry may be buffered by water percolating through carbonate sediments (Santos *et al.,* 2011). On inshore reefs, the metabolism of organically enriched sediments releases additional carbon dioxide during the night. This may additionally reduce pH and increase partial pressure of carbon dioxide, which is indeed observed in currently ongoing field measurements (Uthicke, pers. comm.), potentially affecting small organisms such as coral juveniles and crustose coralline algae that live within the boundary layer around such organically enriched sediments. River runoff may therefore further aggravate the effects of ocean acidification on inshore coral reefs (Kleypas *et al.,* 2001; Anthony *et al.,* 2011).

Conclusion

The last ten years of research and monitoring have added further new and strong evidence to conclude that the exposure to high levels of nutrients, sediments and turbidity from the terrestrial runoff of soils and fertilisers alters the physiology of inshore corals. This has changed inshore coral reef communities, including making them more vulnerable to the impacts of climate change. Reefs in the central and southern Great Barrier Reef have lost over half of the coral cover, with recent losses also in the inshore zone. A large proportion (42 per cent) of the Great Barrier Reef-wide coral cover loss is attributed to predation by crown-of-thorns starfish, with particularly severe declines in the central and southern sections, where terrestrial runoff and other forms of human use are greater than in the far north. There is now strong evidence that flood-derived nutrients contribute significantly to the formation of crown-of-thorns starfish population outbreaks, which affect the Great Barrier Reef coral reefs way beyond the direct reach of the flood plumes. Furthermore, all agricultural runoff in the major catchments contains measurable and sometimes toxic concentrations of herbicides which may exacerbate the pressures on inshore coral communities. The available evidence indicates that a reduction in catchment pollutant loads is essential to halt and reverse further decline in Great Barrier Reef ecosystem condition at a time of rapidly warming climate and progressive ocean acidification.

Seagrass ecosystems

Seagrass meadows are a critical component of the Great Barrier Reef inshore ecosystems, supporting charismatic megafauna such as dugong and turtles, and the productivity of adjacent ecosystems including coral reefs and mangroves (Waycott *et al.,* 2011). Approximately 3063 square kilometres of inshore seagrass meadows (less than 15 metres depth) have been mapped within the Great Barrier Reef World Heritage Area waters since 1985 (McKenzie *et al.,* 2010). The greatest area of seagrass in the Great Barrier Reef World Heritage Area (31,778 square kilometres) is located in the deeper waters (more than 15 metres) of the lagoon, however these meadows are relatively sparse, composed of only a few species and not well documented (McKenzie *et al.,* 2010; Coles *et al.,* 2009). Of greatest conservation concern are the seagrass meadows adjacent to the developed coast, which are at risk from a wide diversity of impacts and are especially vulnerable to land-based threats, including agricultural runoff and coastal development (Grech *et al.,* 2011). The shallow inshore waters where seagrasses proliferate and are easily accessible are the main foraging areas for dugong and sea turtle (Marsh *et al.,* 2011) and the major nursery areas for important Great Barrier Reef fisheries (Coles *et al.,* 2007).

Long term monitoring since 1999 indicates that inshore seagrass meadows along the Great Barrier Reef developed (agricultural/urban) coast have been declining over the last three to five years and are currently in a poor condition (McKenzie *et al.,* 2012a, 2012b; www.seagrasswatch.org, accessed 13 January 2013). Indicators are declining trajectories in seagrass abundance, reduced meadow extent, decreasing reproductive effort, limited or absent seed production and increased epiphyte loads at most locations (McKenzie *et al.,* 2012a). Multiple stressors are implicated in this decline, the most significant being light limitation (from increased turbidity) and nutrient enrichment (predominately nitrogen) at the majority of locations and over the long-term (McKenzie *et al.,* 2010; Mellors *et al.,* 2005). Increased epiphyte loads, possibly stimulated by nutrient loading, contribute to light limitation and compromise the ability of the plants to photosynthesize. Losses were exacerbated by climatic episodic events, as demonstrated by the widespread and substantial losses documented after the floods and cyclones of early 2011 (McKenzie *et al.,* 2012a).

Water quality and seagrass community composition

Seagrasses require light, nutrients, carbon dioxide, substrate for anchoring, and tolerable salinity, temperature and pH to survive. There are many drivers of seagrass distribution, abundance and diversity in the Great Barrier Reef, resulting in complex and dynamic seagrass habitats (Waycott *et al.,* 2005; Collier and Waycott 2009). Physical factors, such as tidal variation, sediment type and wave exposure, are the overarching factors influencing seagrass distribution (Grech and Coles 2010). However, colonization depth and abundance are generally controlled by water quality (light penetration and nutrient concentrations) (Abal and Dennison 1996; Waycott and McKenzie 2010; Collier *et al.,* 2012a; McKenzie *et al.,* 2012a).

Light – turbidity

Water quality affects seagrass through its various biological, chemical and physical characteristics. Parameters that reduce light availability will have the greatest impact, as it is the most dominant overriding factor in seagrass growth. Chronic and intermittent increases in turbidity (Schaffelke *et al.,* 2012b), reduce light available for photosynthesis resulting in “senescence” (shedding of leaves on shoots, thus reducing abundance) and eventual plant death (Collier *et al.,* 2012a). Burial of seagrass during sediment deposition can also occur when seagrass meadows grow close to river mouths with large sediment discharge (Campbell and McKenzie 2004; Cabaço *et al.,* 2008) or in close proximity of dredging activities (Pringle 1989; Erftemeijer and Lewis III, 2006). Seagrasses can survive below their minimum light requirements (the amount of light needed for long-term survival) for short periods of time (weeks to months), but with reduced abundance and productivity. Event-based light thresholds (two weeks to three months tolerance of reduced light) have been derived for northern Great Barrier Reef seagrass meadows dominated by *Halodule uninervis* (Collier *et al.,* 2012b), and southern seagrass meadows dominated by *Zostera muelleri* (Chartrand *et al.,* 2012). Deeper water seagrasses are low-light adapted and occur to depths of 60 metres (Coles *et al.,* 2009), however suspended sediments, plankton and coloured dissolved organic matter can reduce not only the quantity but also the spectral quality of light reaching such depths (Kirk 2011).

Nutrients

When sufficient light is available for photosynthesis, the primary limiting nutrients for seagrass growth and production are nitrogen and phosphorus (Udy *et al.,* 1999). Dissolved inorganic nutrients are the most readily absorbed form of nutrients by both the roots and the leaves of seagrasses (Romero *et al.,* 2006). Sources of dissolved inorganic nutrients and dissolved inorganic phosphorus for inshore seagrasses are, therefore, land runoff and the biogeochemical transformation processes of organic nutrients in the sediment and water column. While additional nutrients can enhance seagrass growth (Udy and Dennison 1997a; Udy *et al.,* 1999), elevated water column nutrients can favour the growth of plankton, macroalgae and epiphytic algae, all of which attenuate light availability to seagrass leaves. Toxic effects of water column nitrogen concentrations can occur if nitrate and ammonium concentrations reach approximately 200 micrometres (Burkholder *et al.,* 1992; van Katwijk *et al.,* 1997; Brun *et al.,* 2002). However, sensitivity to nutrient enrichment appears to be species-specific (Ralph *et al.,* 2006) and differs between seasons, with greater impacts during senescent months (e.g. winter) (Brun *et al.,* 2002) or when light availability is low and temperature is high (van Katwijk *et al.,* 1997). Seagrasses exposed to high nutrient concentrations have high tissue concentrations of nutrients and amino acids (Udy and Dennison 1997b) which can be used as early-warning indicators of potential broad-scale seagrass declines. Although current nutrient concentrations in the Great Barrier Reef seem to have not yet reached critical levels for seagrasses (Waycott *et al.,* 2005), there are indications of long-term increases in seagrass leaf tissue nitrogen concentrations, increasing epiphyte loads, and reduced light availability, since records began in 2005 (McKenzie *et al.,* 2012a).

Pesticides

Inshore seagrass meadows of the Great Barrier Reef are also exposed to herbicides (Haynes *et al.,* 2000; Lewis *et al.,* 2009), in most cases to mixtures of compounds (Lewis *et al.,* 2012a). The photosystem II inhibiting herbicide diuron, which is commonly found in Great Barrier Reef inshore waters, adversely affects seagrass productivity at concentrations measured in flood plumes (Haynes *et al.,* 2000; McMahon *et al.,* 2005). The contribution of herbicides to recent widespread seagrass losses is unknown. However, very high concentrations of diuron and atrazine may have contributed to localised losses, such as in Sarina Inlet (McKenzie *et al.,* 2012b; Kennedy *et al.,* 2012a). Species sensitivity to herbicides is unclear and possibly regionally different, and acclimation to chronic sub-lethal concentrations has been suggested (Ralph *et al.,* 2006). While acclimation can occur, the 'metabolic cost' hypothesis (Calow and Silby 1990) states that the metabolic changes necessary for acclimation will lead to increased energy consumption resulting in adverse effects on reproduction and growth. Such trade-offs between various biological traits have been observed in organisms experiencing stimulation of one particular trait (e.g. Stearns, 1989; Rose *et al.,* 2000). Recent research on benthic microalgae exposed to multiple herbicides suggests additive toxicity (Magnusson *et al.,* 2010) but the responses of seagrasses, especially to chronic exposure are unknown.

System-wide flow on effects of seagrass decline

The responses of seagrasses in the Great Barrier Reef to chronic changes in water quality are poorly understood due to a limited research effort. Most research has focused on acute water quality impacts as a result of dredging operations or flood impacts. It is likely that chronic declines in water quality can shift seagrass meadow distributions and result in changes to community composition (Fourqurean *et al.,* 2001). Further, it can reduce their resilience, leaving seagrass populations vulnerable to episodic disturbances, from which the seagrass meadow may fail to recover after severe loss (van Katwijk *et al.,* 2010).

The loss of seagrass from reduced water quality and physical disturbance, as a result of floods and cyclones, is known to have significant flow-on effects to the dugong and green turtle populations, which are highly dependent on the local seagrass meadows that provide their primary food supply (Preen *et al.,* 1995; Preen and Marsh 1995). Malnutrition is known to make the animals prone to disease and other pre-existing conditions and injuries or force the animals to move long distances to find alternative food sources. As a consequence of the widespread loss of seagrass along the east coast of Queensland in early 2011, stranding rates of turtles and dugong increased during that year across the Great Barrier Reef (581 and 101 individuals respectively; data courtesy of StrandNet, accessed 17 January 2013). These were the highest mortality rates since records commenced in 1998. Where seagrass meadows are unable to recover, the reproductive potential of dugongs and green turtles is likely to be reduced for many years, which will have ongoing population level impacts, limiting their ability to recover. If there are further losses of seagrass meadows, populations of dugongs and green turtles will continue to decline until levels match the new carrying capacity of the remaining seagrass habitats. The flow-on effects of seagrass loss to other associated fauna or fisheries is less obvious in subtropical and tropical systems, as it may not immediately manifest or cause community shifts rather than losses. For example, declines in one species of penaeid prawn from seagrass loss were balanced by an increase in another species of penaeid prawn or finfish (Connolly *et al.,* 1999), but it is questionable whether they would present a similar economic value. The consequences of seagrass loss are not only limited to associated fauna, but as environmental engineers, declines in seagrass can have broader consequences related to coastal processes, such as carbon capture and nutrient dynamics, sediment stabilisation, and habitat connectivity (Waycott *et al.,* 2007).

Conclusion

The capacity of seagrass meadows to naturally recover community structure following disturbance will depend on the interaction between light availability, nutrient loads, suitable habitat and the availability of seeds to form the foundation of new populations (McKenzie *et al.,* 2012a). Recovery is known to be species-dependent and is expected to take between two and eight years without further disturbances (McKenzie *et al.,* 2012a). At present, Great Barrier Reef seagrass meadows appear to have variable recovery potential due to varying light levels, reproductive effort and seed availability both spatially and temporally. Recovery processes have been key to maintaining the long term health of seagrass meadows. Unfortunately, the characteristics of seagrass meadows that confer resilience are not well understood, and require priority attention as we are challenged to maintain long-term resilience of these important ecosystems as other cumulative impacts worsen due to changes in disturbance regimes (see above).

Mangrove ecosystems

It is estimated that the current area of mangroves at the Great Barrier Reef wide scale is 98 per cent of the pre-clearing extent (Wilson and Pennay, 2012), however, their condition is largely unknown. There have been reports of both expansion (e.g., Burdekin region) and localised die-back of mangrove stands (e.g., Mackay Whitsundays), with the underlying drivers being complex and site-specific. Few studies have focused on the effects of changing water quality on mangroves in the Great Barrier Reef region (e.g., Woodroffe 1992; Wolanski 1992; Lovelock *et al.,* 2009). However, other research that is transferrable to the Great Barrier Reef showed mangrove forest structure and function is driven by the interplay between topography, climatic factors and the delivery of nutrients, sediments, freshwater and other contaminants to the coastal inshore area.

Sedimentation

Significant new mangrove stands and landward range expansion have been reported in some areas of the Great Barrier Reef in recent decades. These changes were correlated with increased sedimentation due to vegetation clearing in adjacent catchments and, in the case of the Fitzroy River, changes in hydrology due to the construction of a major river barrage (see review in Schaffelke *et al.,* 2005 and references therein). Dredging activities have also been linked to increases in mangroves on Magnetic Island (Duke and Wolanski 2001). However, excessive sedimentation may bury seedlings and pneumatophores, reduce growth and cause mortality of mangroves (e.g., around Bowen; Ellison 1998).

Nutrients

Mangrove ecosystems are commonly limited by nitrogen and/or phosphorus, and nutrient enrichment can enhance mangrove growth and reproduction (Boto and Wellington 1983; Clough 1983; Feller 1995; Feller *et al.,* 2002 and 2003; Lovelock and Feller 2003; Lovelock *et al.,* 2004, 2006a,b and 2007; Martin 2007; Feller *et al.,* 2009; Martin *et al.,* 2010). However, the benefits of increased mangrove growth in response to increased nutrient levels is offset by decreased resilience and higher rates of mortality during periods of drought, with mortality increasing along gradients of increasing salinity and aridity (Lovelock *et al.,* 2009). This is because increases in nutrients (particularly nitrogen) stimulates the growth of shoots relative to roots (Gleeson 1993; Naidoo 2009), which may enhance productivity under favourable conditions, but increase the vulnerability of mangroves to water stress (Martin 2007; Martin *et al.,* 2010; Lovelock *et al.,* 2009). Less root biomass can also enhance subsidence of soils relative to the level of the sea, possibly decreasing resilience with increasing sea level (McKee *et al.,* 2007).

Nitrogen and phosphorus differentially affect patterns of photosynthetic carbon gain and water use in mangroves, with nitrogen enrichment resulting in higher water-use efficiency and phosphorus enrichment resulting in lower water-use efficiency under highly saline conditions (Lovelock and Feller 2003; Martin 2007; Martin *et al.,* 2010). Hence nutrient enrichment can significantly alter mangrove forest structure and function along salinity gradients (Martin *et al.,* 2010) with far reaching consequences under present and future climatic conditions where environmental variability is predicted to increase (Lough 2011). The positive effects of phosphorus on water uptake and transport processes (Lovelock *et al.,* 2006a; Lovelock *et al.,* 2006b; Martin 2007) could influence the probability of survival during periods of water stress. This is supported by studies in the Great Barrier Reef where canopy loss was significantly greater in trees fertilised with nitrogen during a period of very low rainfall at the most saline site (Lovelock *et al.,* 2009). The emergence of a pattern in tree mortality with nutrient enrichment across species and biogeographic regions highlights the importance of climatic interactions in determining how coastal ecosystems respond to eutrophication (Lovelock *et al.,* 2009).

Given the response to nutrients across species and bioregions (Lovelock *et al.,* 2009), high nutrient loads delivered in flood waters to the Great Barrier Reef (Brodie *et al.,* 2010) could be a contributor to the localised dieback of mangrove forests when followed by periods of above-average temperatures (Lovelock *et al.,* 2009; Martin *et al.,* 2010) or low rainfall (Eslami-Andargoli *et al.,* 2010). Hence, although mangroves may have a relatively high capacity for nutrient retention and can purportedly protect the marine environment from land-derived nutrient pollution (Valiela and Cole, 2002; Alongi and McKinnon, 2005; Adame *et al.,* 2010), nutrient enrichment can have negative consequences for mangrove forests in the long term and their capacity for retention of nutrients may be limited (Adame *et al.,* 2010; Reef *et al.,* 2010).

Pesticides

The widespread dieback of mangroves in the Mackay region from the mid-1990s to the early 2000s (Jupiter *et al.,* 2007) was initially hypothesised to be associated with levels of diuron and other herbicides present in porewater and sediments of the mangrove forests (Duke *et al.,* 2005; Duke 2008). However, this has been disputed due to conflicting evidence and insufficient data (Kirkwood and Dowling 2002; Wake 2005 and 2006; Abbot and Marohasy 2011). Nevertheless, the presence of photosystem II inhibiting herbicides in mangrove habitats is of concern. In laboratory studies, four species of mangrove seedlings showed significant reduction in photosynthetic efficiency after acute herbicide exposure to roots (Bell and Duke 2005). However, experimental concentrations of diuron were many times higher than the concentrations detected in sediments and porewater of the Pioneer River. Herbicides may persist in mangrove environments due to low rates of porewater flushing and microbial degradation, and long-term studies of chronic exposure of mangroves to herbicides and other co-occurring factors at biologically relevant concentrations are urgently needed.

Other environmental drivers

Global studies indicate that attributing mortality in mangrove stands to a single factor is overly simplistic. It is likely that a combination of climatic factors such as small scale changes in evaporative demand or hydrology due to changes in rainfall (Eslami-Andargoli *et al.,* 2010) and acute pressures (e.g. the burial of pneumatophores or freshwater inundation following flood events) on top of the chronic stress of poor water quality (e.g. high nutrient and pesticide loads) are driving localised dieback of mangrove forests in the Great Barrier Reef.

Changes in the frequency and intensity of rainfall will have major positive and negative effects on the species composition of coastal ecosystems, both directly and indirectly, through modulation of porewater salinity, humidity, nutrient availability and rates of sedimentation. For example, increased rainfall can enhance productivity by creating conditions that are favourable for plant physiological function (Ball 1998), while prolonged inundation may 'suffocate' mangroves (McKee 1996). Recent evidence from Florida indicates that after hurricanes, primary production of mangrove forests can be significantly reduced for years (Barr *et al.,* 2012), which supports observations in northern Queensland following Tropical Cyclone Yasi and the associated flooding. A significant positive relationship was demonstrated between median rainfall and landward expansion of mangroves into salt marsh habitat in Moreton Bay (Eslami-Andargoli *et al.,* 2010) and the Hinchinbrook region (Ebert 1995). Conversely, a long-term decline in rainfall in the Fitzroy catchment was related to a shift in vegetation from mangrove dominated communities to salt marsh (Duke *et al.,* 2003).

There is consensus that rising sea level will affect coastal ecosystems in Australia, despite limited monitoring data (Lovelock *et al.,* 2011a; Lovelock *et al.,* 2012). However, mangroves are unlikely to keep pace with projected increases in the rate of sea level rise by 2100, although there is a lack of information on species' adaptive capacity and their ability to migrate landward may be restricted by natural barriers or human structures.

Cyclones can have acute destructive impacts on mangrove stands (e.g. defoliation, abrasion, stem breakage, uprooting and smothering by sediment) (Woodroffe and Grime 1999; Paling *et al.,* 2008). However, cyclones can also increase post-disturbance growth rates through alleviation of hyper-salinity and increases in the availability of nutrients (Lovelock *et al.,* 2011b). Quantitative data from Australia are rare and there are few accounts documenting large-scale damage to mangrove stands in the Great Barrier Reef region (e.g. Hopley 1974; Houston 1999). Responses to canopy damage (Santini *et al.,* 2012) and vulnerability to severe storms (Baldwin *et al.,* 2001) are generally species-specific. Global studies on various mangrove forest types indicates that mangroves can generally recover from storm damage if patches of reproductive trees remain and there are no major changes in hydrology and geomorphology to prevent re-establishment (Ellison 1998; Sherman *et al.,* 2001). However, predicted increases in the frequency and intensity of extreme weather events may alter the community structure and productivity of mangrove forests in the Great Barrier Reef region to favour species more tolerant to disturbances with rapid potential for regeneration.

Conclusion

The responses of mangrove forests to poor water quality are complex and at times appear contradictory, because the responses are strongly dependent on interactive effects between multiple direct and indirect factors, and local climatic conditions. For example, increased freshwater input and sedimentation within wetlands may increase the habitat area available for colonisation, and nutrient enrichment can have positive influences on plant growth. However, increases in productivity and growth expected under high nutrient levels are likely to be offset by increased mortality rates and canopy loss during periods of drought or heightened cyclonic and storm activity. The impact of eutrophication on mangrove forest structure and function is likely to become more severe along gradients of increasing salinity and aridity, with potential implications for the community composition of the ecosystem and the ecosystem services provided (e.g. nursery function, carbon sequestration).

The complexity in the responses of mangroves to run-off presents many challenges in managing coastal ecosystems at the local scale. However, the emergence of a pattern of increased susceptibility to environmental variability across species and biogeographic regions underscores the importance of climatic interactions with the intertidal landscape in determining how coastal ecosystems respond to eutrophication and indicates that improvements in water quality will help to maintain the resilience of communities in a changing climate.

Coastal freshwater wetlands

Wetlands have an important role in the landscape and reef, providing habitat and vital ecosystem processes. Freshwater wetlands exist across the Great Barrier Reef catchments. Loss of wetlands is a key indicator of wetland status. The area of non-riverine freshwater wetlands remaining in Great Barrier Reef catchments in 2009 equates to 78 per cent of the pre-European extent (280, 000 hectares) (Department of the Premier and Cabinet 2013). Seventy-seven per cent (262,000 hectares) of palustrine (swamps) and 97 per cent (21,000 hectares) of lacustrine (lakes) wetlands remained in 2009 (Department of the Premier and Cabinet 2013). However, this result which summarises wetland extent across the <u>whole</u> Great Barrier Reef region masks any variations in wetland loss in <u>parts</u> of the catchment (Department of the Premier and Cabinet 2013). For example, 53 per cent of vegetated freshwater swamps have been lost from the whole Herbert River catchment compared to an 83 per cent loss over the lowland parts of the catchment (Department of the Premier and Cabinet 2013; Johnson *et al.,* 1999).

In addition to loss, many wetlands in the Great Barrier Reef catchments are under threat (Brodie and Mitchell 2005), suffering a range of impacts associated with loss of connectivity, sediment and nutrient overload and weed infestations (Great Barrier Reef Marine Park Authority 2012) as, for example, shown in the Wet Tropics (Tsatsaros *et al.,* 2013). This can lead to a loss of ecosystem function, for example, when a wetland is exposed to high sediment loads, it may no longer function as a sink (Wetzel 2001) resulting in the export of pollutants (Verhoeven *et al.,* 2006) to other waterbodies. Especially during wet season floods, large amounts of nutrients, sediments and pesticides can be mobilised and transported downstream (Brodie *et al.,* 2012; Joo *et al.,* 2012).The rapid passage of water reduces water residence time in freshwater wetlands and many wetland processes such as carbon metabolism, sedimentation and denitrification can be compromised (Brodie and Mitchell 2005).

Limited research has been undertaken on the impacts of poor water quality on freshwater wetlands in the Great Barrier Reef catchments (Brodie *et al.,* 2012; Arthington *et al.,* 1997). Consequently, this review includes relevant research from other places.

Sedimentation and weed infestation

The rate of sediment accumulation in a stream is dependent on the balance of erosion, deposition, re-suspension and re-deposition (Arthington *et al.,* 1997). This balance may be changed by the growth of vegetation, which traps suspended particles, causing aggradation and further plant colonisation (Arthington *et al.,* 1997).

Many tropical lowland streams in far north Queensland are congested with weeds as well as sediment accumulated from cropland erosion (Bunn *et al.,* 1998). Weed growth is increased in nutrient-rich water and a higher biomass can decrease the rate of water flow causing siltation and a reduction of water depth (Howard and Harley 1998). Connolly *et al.,* (2012) observed that alterations to wetland hydrology from crop irrigation changed the seasonal growth patterns of native wetland plant species causing widespread proliferation of exotic weed species in many wetlands. Perna *et al.,* (2012) indicate that once weeds are established in surface waters, water quality is further degraded by a reduction in oxygen levels through the decomposition of organic matter, with significant negative effects on aquatic species such as fish and macro-invertebrates. Many coastal wetlands in the Great Barrier Reef catchment area are now water-logged, have poor water quality and are infested with weeds creating chronic anoxic conditions that affect the connectivity between wetlands (Connolly *et al.,* 2012; Perna *et al.,* 2012).

The mobilisation and transport of accumulated sediment downstream is a major effect of flood events. Bunn *et al.,* (1998) noted that a 38 kilometre section of the lower Johnstone River at one stage was choked with para grass and estimated that 760 000 tonnes of sediment was trapped in this weed and was available for mobilisation during a flood. A case study from Healeys Lagoon (Queensland Wetlands Program), Burdekin Shire, describes how a sudden flood event flushed out almost the entire biomass from a weed infested lagoon. Weed mats have also been observed to cause damage to infrastructure such as bridges during floods (Connolly *et al.,* 2012).

Suspended sediments can lower water temperature by increasing reflection (Ryan 1991), reduce light availability to aquatic plants and impede fish foraging (Bruton 1985; Wetzel 2001). Suspended particles have also been observed to clog the gills of fish therefore adversely impacting gas exchange and possibly leading to death (Ryan 1991). At Tinaroo Dam in Far North Queensland, sedimentation smothered the eggs and larvae of benthic macro-invertebrates and limited the availability of food to grazers (MacKinnon and Herbert 1996).

Nutrients

Northern Australian waterbodies including freshwater wetlands are likely to be affected by changes to nutrient inputs (Brodie and Mitchell 2005; Tsatsaros *et al.,* 2013). The effect of nutrients on wetlands will depend on the wetland type, with some being more sensitive than others (Reddy and D'Angelo 1997). Nutrients such as nitrogen and phosphorus occur naturally and are an essential part of aquatic food chains (Arthington *et al.,* 1997). When nutrients are elevated beyond a critical threshold, drastic changes can suddenly occur. These include impacts on the metabolism and growth of organisms (Wetzel 2001) leading to shifts in species dominance and composition, changes in ecosystem functioning such as carbon and nutrient outputs, trophic interactions and/or nutrient cycling rates (Verhoeven *et al.,* 2006). Congdon and Lukacs (1996) suggest that nutrient inputs during the dry season are likely to have a greater impact on wetlands in the Burdekin River area as algal blooms are more common during times of low flow.

In freshwater wetlands, nutrient enrichment of water often facilitates growth of exotic weed species (Brodie and Mitchell 2005) and floating weed mats (Herbert *et al.,* 1996). Floating aquatic weeds typically flourish in tropical and sub-tropical locations (Howard and Harley 2008). Weed growth is stimulated by inflow of nutrient rich water, such as runoff from tropical rains, particularly into static waters (Howard and Harley 1998). Some species of weed such as water hyacinth and *Salvinia* can develop dense mats that significantly reduce light penetration into the water, thereby inhibiting photosynthesis in submerged plants causing reduced oxygen and increased carbon dioxide levels. This can have catastrophic effects on aquatic fauna, e.g. leading to fish kills (Howard and Harley 1998). Floating plants such as water hyacinth and emergent grasses such as para grass have been observed to cause severe degradation in northern Australian wetlands in close proximity to agriculture (Arthington *et al.,* 1997)

Pesticides

Pesticide residues, especially of photosystem II-inhibiting (PSII) herbicides are reportedly widespread across the Great Barrier Reef catchment area (Davis *et al.,* 2008, in press; Smith *et al.,* 2012). However, pesticides are composed of many different compounds with different toxicity and mode of action (Finlayson and Silburn 1996) and their impact on Australian tropical freshwater and estuarine aquatic organisms is largely unknown (Magnusson *et al.,* 2010). From research undertaken on flood plumes in the Tully-Murray River (Bainbridge *et al.,* 2009) and in Barratta Creek in the lower Burdekin (Davis *et al.,* 2012) we conclude that the frequent detection of diuron, atrazine and hexazinone residues in waterways draining sugarcane indicates a risk to downstream freshwater and marine ecosystems. Different combinations of photosystem II inhibiting herbicides produce additive toxic effects in tropical benthic microalgae, causing a significant reduction in growth (Magnusson *et al.,* 2010). Given the sensitivity of tropical marine microalgae to photosystem II-inhibiting herbicides (ibid.) it is reasonable to assume that herbicides would also inhibit photosynthesis of macrophytes and microalgal species in freshwater and estuarine wetlands.

Conclusions

Research suggests that poor land use and management practices are having an adverse impact on freshwater coastal wetlands in Great Barrier Reef catchments as well as in the marine receiving waters. This is particularly evident for sediment and nutrient inputs and irrigation techniques that impact wetland structure and function by facilitating weed growth; loss of connectivity, reduced oxygen levels and flow rate, which contribute to loss of biodiversity and further degradation of the water quality reaching the Great Barrier Reef lagoon. Recent evidence shows that the Bowling Green Bay Ramsar site (mostly marine wetlands) and the adjacent freshwater wetlands listed in A Directory of Important Wetlands in Australia (Environment Australia 2001) are at high risk of damage due to pesticide loading.

The key measure of wetland efficacy at present is extent, although a tool is currently being developed under the Queensland Wetland Program to also examine wetland condition. There are many indicators to show that the health of freshwater ecosystems is already impaired by inappropriate agricultural land use, hydrological change, riparian degradation and weed infestation. The full effect of poor water quality on freshwater wetland ecosystems, however, is poorly understood due to limited research.

Overall conclusions

The scientific consensus presented in this review confirms (i) that marine water quality continues to be negatively affected by the discharge of excess nutrients, fine sediments and pesticides from the adjacent catchments and (ii) that poor marine water quality is a major cause for the currently poor state of many of the key marine ecosystems of the Great Barrier Reef. The recent series of extreme weather events has highlighted the importance of climate drivers affecting the variability of environmental conditions between years, including marine water quality. Extreme events, especially tropical cyclone and major river floods appear to perturb the dynamics of Great Barrier Reef ecosystems for extended periods of time, long after the acute pressures have subsided.

Research published since 2008 strengthens the original conclusions, as relevant to this chapter, of the previous Scientific Consensus Statement, that:

1. Water discharged from rivers to the Great Barrier Reef continues to be of poor quality in many locations, especially during major flood events.
2. Land-derived suspended sediments, nutrients and pesticides are present in the inshore Great Barrier Reef lagoon at concentrations likely to cause environmental harm.
3. There is strengthened evidence of the causal relationship between water quality and the health of coastal and marine ecosystems.
4. Environmental variability caused by climate change is expected to exacerbate the effects of continued runoff of excess suspended sediments, nutrients and pesticides with negative consequences for Great Barrier Reef ecosystem health.
5. Improvements in marine water quality and associated marine ecosystem condition are likely to be slow and difficult to detect because of the highly variable baseline, strong inter-annual rainfall variability, lags in ecosystem responses and potentially long recovery periods.

Lines of evidence

Exposure of the Great Barrier Reef lagoon to land runoff

- Water from catchment runoff carrying excess loads of nutrients and fine suspended sediments are carried further into the Great Barrier Reef lagoon than previously thought.
- Any increase in the frequency of extreme weather events that influence the loads of contaminants exported to inshore waters will increase the risk of exposure and reduce the resilience of Great Barrier Reef ecosystems.
- Land-based pollutants are exported into the Great Barrier Reef lagoon during short periods of days to weeks per year during river floods, but they remain biologically active for long periods (years to decades) and are likely to produce adverse effects on Great Barrier Reef ecosystems for long periods.
- Elevated concentrations of nutrients (both dissolved and particulate) and of suspended solids are observed in costal and inshore waters after exposure to riverine flood plumes and after resuspension of bottom sediments by strong winds or tidal currents.

Condition of Great Barrier Reef lagoon water quality

- Seasonality is the strongest determinant of water quality in the Great Barrier Reef lagoon. There are also distinct cross-shelf gradients with mean concentrations of water quality variables being 0.6- to five-fold higher in the coastal and inshore areas compared to the offshore lagoon, especially adjacent to the Wet Tropics and Burdekin regions in the central Great Barrier Reef.

- Physical forcing by wind, waves and tides are the main drivers of turbidity in the inshore Great Barrier Reef lagoon. Importantly, turbidity is also positively correlated with river flow, with the relationship stronger at sites close to a river mouth, highlighting the influence of the fine suspended sediments loads carried by the rivers on the inshore marine environment. Turbidity remains elevated around many inshore reefs for weeks to months after periods of river flow.
- A long-term water quality data series showed clear relationships between land-use change on the catchment (land clearing rate) and marine water quality. Seasonality was most important, however, high catchment vegetation clearing rates, together with high river discharge were associated with high concentrations of several water quality variables in the adjacent coastal marine waters.
- Nutrients from land runoff are rapidly taken up by phytoplankton and the resulting organic matter is cycled through the marine food web and transformed e.g. into detrital (“marine snow”) particles that may be deposited on benthic communities. By cycling through the food web, event-driven inputs of nutrients are likely to not only have an acute but also a longer-term ecosystem effect.
- Herbicides are detectable in the inshore Great Barrier Reef lagoon and in estuarine waters throughout the year, generally with higher concentrations in the wet season and closer to the coast. While herbicide concentrations do not exceed water quality guidelines during most of the year, the consequences for marine life of low level chronic exposure to pesticide mixtures or in combination with abiotic factors such as elevated turbidity or nutrients or climatically induced factors are unknown.

Condition of Great Barrier Reef coral reef ecosystems and response to water quality

- Coral cover over large areas of the central and southern Great Barrier Reef has been in decline for several years. The causes of coral loss vary from reef to reef, depending on exposure to tropical cyclones, outbreaks of the crown-of-thorns starfish or coral disease, temperature anomalies causing coral bleaching, and floods. The persistence of coral communities is reliant on high resilience, i.e. on maximising both resistance to disturbances and ability to recover during periods without disturbance.
- Poor water has the potential to both increase the susceptibility of corals to disturbances and then suppress their subsequent recovery.
- Inshore coral reefs are directly and negatively affected by increased turbidity and sedimentation, especially after extreme weather events leading to river floods.
- Coral reefs are also indirectly affected by inputs of excess nutrients, especially after large flood events. Increased phytoplankton productivity and biomass in the lagoonal water column increases the likelihood of outbreaks of the crown-of-thorns starfish as it greatly enhances the survival of crown-of-thorns starfish larvae.
- In the past years with extreme weather events, the incidence of coral disease has increased proportionally with the discharge of local rivers.
- Water turbidity and the proportion of fine-grained particles in the reef sediments have also increased during the period of increasing river discharge, reducing coral cover and recruitment, most likely due to light limitation and smothering.
- Several foundation species are highly sensitive to acute exposure of herbicides, and experiments suggest that herbicides increase the vulnerability of Great Barrier Reef species to the negative effects of climate change.

Condition of Great Barrier Reef seagrass ecosystems and response to water quality

- Inshore intertidal seagrass meadows along the Great Barrier Reef developed coast have been declining over the last three-five years and are currently in a poor condition, based on assessments of abundance, meadow size, reproductive effort and epiphyte load.
- Although current nutrient concentrations in the Great Barrier Reef seem to have not yet reached critical levels for seagrasses, there are indications of long-term increases in seagrass leaf tissue nitrogen concentrations, increasing epiphyte loads, and reduced light availability, since records began in 2005.
- There are recent widespread seagrass losses in the Great Barrier Reef. The contribution of herbicides to these losses is unknown, but it is likely that high diuron and atrazine concentrations have contributed to localised losses.
- The loss of seagrass from reduced water quality, as a result of floods, together with cyclones, have significant flow-on effects to the dugong and green turtle populations which are highly dependent on the local seagrass meadows as their primary food supply. Hundreds of dugong and turtles have been found dead and washed onto beaches after such cyclones in recent years or have been forced to move long distances to find alternative food sources.

Condition of Great Barrier Reef mangrove ecosystems and response to water quality

- The responses of mangroves to poor water quality are complex and poorly understood.
- Increased sedimentation, e.g. due to upstream vegetation clearing or marine dredging activities, has led to expansion of some mangrove stands. However, excessive sedimentation may also bury seedlings and pneumatophores, reduce growth and cause mortality of mangroves.
- Higher nitrogen availability favours growth of mangrove shoots relative to roots, enhancing productivity under favourable conditions, but increases vulnerability to water stress. Hence the benefits of increased growth in response to increased nutrient levels is offset by decreased resilience to environmental variability, and nutrient-induced mortality is greater at sites subject to periods of low rainfall, low humidity and high sediment salinity.

Condition of freshwater wetlands and response to water quality

- The area of non-riverine freshwater wetlands remaining in Great Barrier Reef catchments in 2009 equates to 78 per cent of the pre-European extent (283, 050 hectares). Seventy-seven per cent (261, 820 hectares) of palustrine (swamps) and 97 per cent (21 230 hectares) of lacustrine (lakes) wetlands remained in 2009.
- Poor land use and management practices are having an adverse impact on the remaining coastal freshwater wetlands, with sediment and nutrient inputs and certain irrigation techniques affecting wetland structure and function by facilitating weed growth, loss of connectivity, reduced oxygen levels and flow rate that contribute to loss of biodiversity and further degradation of the water quality reaching the Great Barrier Reef lagoon.
- The effects of poor water quality on freshwater wetland ecosystems are not well understood due to limited research. However, recent evidence shows that the Bowling Green Bay Ramsar site (mostly marine wetlands) and the adjacent freshwater wetlands listed in A Directory of Important Wetlands in Australia (Environment Australia 2001) are at high risk of damage due to pesticide loading.

Key knowledge gaps

While the impacts of poor water quality on corals and seagrass are increasingly understood, less is known of its effects on mangroves, freshwater wetlands or the charismatic fauna such as turtle and dugong that depend on inshore ecosystems. The gaps in our understanding of the response of marine ecosystems to poor water quality hamper our ability to support the selection and prioritisation of management actions and to assess whether management interventions have been successful. In particular, we need to better understand the:

- quantitative links of loads from individual catchments to marine concentrations of nutrients, fine sediments and pesticides
- transport and transformation of nutrients, organic matter and pesticides in the Great Barrier Reef lagoon after flood events (e.g. the remineralisation of terrestrial particulate nutrients, flocculation of fine sediments, fate of plankton blooms, pesticide degradation, etc.)
- responses of key Great Barrier Reef ecosystem components (coral reef biota, seagrasses, mangroves) to chronic low-level exposure to herbicides and/or excess nutrients
- responses of key Great Barrier Reef ecosystem components to multiple flood plumes, which includes quantifying the interactions between salinity, nutrients, turbidity, and pesticides
- characteristics of seagrass meadows and mangroves that maintain long-term resilience under cumulative of poor water quality, disturbance and extreme events
- regionally specific relative roles of riverine inputs, upwelling and water column recycling of nutrients as drivers of phytoplankton biomass that support survival of crown-of-thorns starfish larvae
- larval ecology of crown-of-thorns starfish and the role of nutrition for survival and recruitment.

Reference list

Abal, E.G., Dennison, W.C., 1996. Seagrass depth range and water quality in southern Moreton Bay, Queensland, Australia. Marine and Freshwater Research 47, 763-771.

Abbot, J.,Marohasy, J., 2011. Has the herbicide diuron caused mangrove dieback? A re-examination of the evidence. Human and Ecological Risk Assessment 17, 1077-1094.

Adame, M.F., Virdis, B.,Lovelock, C.E., 2010. Effect of geomorphological setting and rainfall on nutrient exchange in mangroves during tidal inundation. Marine and Freshwater Research 61, 1197-1206.

Alongi, D.M., 2002. Present state and future of the world's mangrove forests. Environmental Conservation 29, 331-349.

Alongi, D.M., McKinnon, A.D., 2005. The cycling and fate of terrestrially-derived sediments and nutrients in the coastal zone of the Great Barrier Reef shelf. Marine Pollution Bulletin 51, 239-252.

Álvarez-Romero, J.G., Devlin, M., Teixeira da Silva, E., Petus, C., Ban, N.C., Pressey, R.L., Kool, J., Roberts, J.J., Cerdeira-Estrada, S., Wenger, A.S., Brodie, J., 2013. A novel approach to model exposure of coastal-marine ecosystems to riverine flood plumes based on remote sensing techniques. Journal of Environmental Management 119, 194-207.

Andutta, F.P., Ridd, P.V., Wolanski, E., 2011. Dynamics of hypersaline coastal waters in the Great Barrier Reef. Estuarine, Coastal and Shelf Science 94, 299-305.

Anthony, K.R.N., Ridd, P.V., Orpin, A., Larcombe, P., Lough, J.M., 2004. Temporal variation in light availability in coastal benthic habitats: effects of clouds, turbidity and tides. Limnology and Oceanography 49, 2201-2211.

Anthony, K.R.N., Connolly, S.R., Hoegh-Guldberg, O., 2007. Bleaching, energetics, and coral mortality risk: Effects of temperature, light, and sediment regime. Limnology and Oceanography 52, 716-726.

Anthony, K.R.N., Fabricius, K.E., 2000. Shifting roles of heterotrophy and autotrophy in coral energetics under varying turbidity. Journal of Experimental Marine Biology and Ecology 252, 221-253.

Anthony, K.R.N., Hoogenboom, M.O., Connolly, S.R., 2005. Adaptive variation in coral geometry and the optimization of internal colony light climates. Functional Ecology 19, 17-26.

Anthony, K.R.N., Hoogenboom, M.O., Maynard, J.A., Grottoli, A.G., Middlebrook, R., 2009. Energetics approach to predicting mortality risk from environmental stress: a case study of coral bleaching. Functional Ecology 23, 539-550.

Anthony, K.R.N., Maynard, J.A., Diaz-Pulido, G., Mumby, P.J., Marshall, P.A., Cao, L., Hoegh-Guldberg, O., 2011. Ocean acidification and warming will lower coral reef resilience. Global Change Biology 17, 1798-1808.

ANZECC and ARMCANZ, 2000. Australian and New Zealand Guidelines for Fresh and Marine Water Quality, Vol. 1. The Guidelines, Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand.

Armour, J.D., Hateley, L.R., Pitt, G.L., 2009. Catchment modelling of sediment,nitrogen and phosphorus nutrient loads with SedNet/ANNEX in the Tully–Murray basin. Marine and Freshwater Research 60, 1091–1096.

Arthington, A.H., Marshall, J.C., Rayment, G.E., Hunter, H.M., Bunn, S.E., 1997. Potential of Sugarcane Production on Riparian and Freshwater Environments. In: *Intensive Sugarcane Production: Meeting the Challenges Beyond 2000*. CAB International, Wallingford, UK.

Ashauer, R., Boxall, A.B.A., Brown, C.D., 2006. Predicting effects on aquatic organisms from fluctuating or pulsed exposure to pesticides. Environmental Toxicology and Chemistry 25, 1899–1912.

Ashauer, R., 2010. Toxicokinetic–toxicodynamic modelling in an individual based context—consequences of parameter variability. Ecological Modelling 221, 1325–1328.

Ashauer, R., Boxall, A.B.A., Brown, C.D., 2007a. New ecotoxicological model to simulate survival of aquatic invertebrates after exposure to fluctuating and sequential pulses of pesticides. Environmental Science and Technology 41, 1480–1486.

Ashauer, R., Boxall, A.B.A., Brown, C.D., 2007b. Simulating toxicity of carbaryl to *Gammarus pulex* after sequential pulsed exposure. Environmental Science and Technology 41, 5528–5534.

Bainbridge, Z.T., Brodie, J.E., Faithful, J.W., Sydes, D.A., Lewis, S.E., 2009. Identifying the land-based sources of suspended sediments, nutrients and pesticides discharged to the Great Barrier Reef from the Tully-Murray Basin, Queensland, Australia. Marine and Freshwater Research 60, 1081-1090.

Bainbridge, Z.T., Wolanski, E., Alvarez-Romero, J.G., Lewis, S.E., Brodie, J.E., 2012. Fine sediment and nutrient dynamics related to particle size and floc formation in a Burdekin River flood plume, Australia. Marine Pollution Bulletin 65, 236-248.

Baldwin, A., Egnotovich, M., Ford, M.,Platt, W., 2001. Regeneration in fringe mangrove forests damaged by Hurricane Andrew. Plant Ecology 157, 149-162.

Ball, M.C., 1998. Mangrove species richness in relation to salinity and waterlogging: a case study along the Adelaide River floodplain, northern Australia. Global Ecology and Biogeography Letters 7, 73-82.

Barr, J.G., Engela, V., Smith, T.J., Fuentes J.D., 2012. Hurricane disturbance and recovery of energy balance, CO2 fluxes and canopy structure in a mangrove forest of the Florida Everglades. Agricultural and Forest Meteorology 153, 54-66.

Bell, A.M., Duke, N.C., 2005. Effects of Photosystem II inhibiting herbicides on mangroves; preliminary toxicology trials. Marine Pollution Bulletin 51, 297-307.

Beman, J.M., Arrigo, K.R., Matson, P.A., 2005. Agricultural runoff fuels large phytoplankton blooms in vulnerable areas of the ocean. Nature 434, 211-214.

Bengtson Nash, S.M., McMahon, K., Eaglesham, G., Muller, J.F., 2005. Application of a novel phytotoxicity assay for the detection of herbicides in Hervey Bay and the Great Sandy Straits. Marine Pollution Bulletin 51, 351-360.

Berkelmans, R., Jones, A.M., Schaffelke, B., 2012. Salinity thresholds of *Acropora* spp. on the Great Barrier Reef. Coral Reefs 31, 1103-1110.

Birkeland, C., 1988. Geographic comparisons of coral-reef community processes. Proceedings of the 6th International Coral Reef Conference 1, 211-220.

Birrell, C.L., McCook, L.J., Willis, B.L., 2005. Effects of algal turfs and sediment on coral settlement. Marine Pollution Bulletin 51, 408-414.

Birrell, C.L., McCook, L.J., Willis, B.L., Harrington, L., 2008. Chemical effects of macroalgae on larval settlement of the broadcast spawning coral *Acropora millepora*. Marine Ecology Progress Series 362, 129-137.

Borell, E.M., Bischof, K., 2008. Feeding sustains photosynthetic quantum yield of a scleractinian coral during thermal stress. Oecologia 157, 593-601.

Borell, E.M., Yuliantri, A.R., Bischof, K., Richter, C., 2008. The effect of heterotrophy on photosynthesis and tissue composition of two scleractinian corals under elevated temperature. Journal of Experimental Marine Biology and Ecology 364, 116-123.

Boto, K.G. and Wellington, J.T. 1983. Phosphorus and Nitrogen Nutrient Status of a Northern Australian Mangrove Forest. Marine Ecology Progress Series 11, 63-69.

Botté, E.S., Jerry, D.R., Codi King, S., Smith-Keune, C., Negri, A.P., 2012. Effects of chlorpyrifos on cholinesterase activity and stress markers in the tropical reef fish *Acanthochromis polyacanthus*. Marine Pollution Bulletin 65: 384-393.

Brando, V.E., Blondeau-Patissier, D., Schroeder, T., Dekker, A.G., Clementson, L., 2011. Reef Rescue Marine Monitoring Program: Assessment of terrestrial run-off entering the Reef and inshore marine water quality monitoring using earth observation data. Final Report for 2010/11 Activities. CSIRO, Canberra. Available at: http://www.gbrmpa.gov.au/resources-and-publications/publications/annual-reef-rescue-marine-monitoring-science-report.

Broderius, S., Kahl, M., 1985. Acute toxicity of organic chemical mixtures to the fathead minnow. Aquatic Toxicology 6, 307-322.

Brodie, J., Dekker, A.G., Brando, V.E., Masters, B., Faithful, J., Noble, R., Rohde, K., 2006. Extent and duration of the algal bloom in the Great Barrier Reef Lagoon following river discharge events in the Mackay Whitsunday Region, Australia. In: Proceedings of the 13th Australasian Remote Sensing and Photogrammetry Conference: Earth Observation – From Science to Solutions. Canberra, 2006.

Brodie, J., De'ath, G., Devlin, M., Furnas, M., Wright, M., 2007. Spatial and temporal patterns of near-surface chlorophyll a in the Great Barrier Reef lagoon. Marine and Freshwater Research 58, 342-353.

Brodie, J., Devlin, M., Haynes, D., Waterhouse, J., 2011. Assessment of the eutrophication status of the Great Barrier Reef lagoon (Australia). Biogeochemistry 106, 281-302.

Brodie, J., Fabricius, K., De'ath, G., Okaji, K., 2005. Are increased nutrient inputs responsible for more outbreaks of crown-of-thorns starfish? An appraisal of the evidence. Marine Pollution Bulletin 51, 266-278.

Brodie, J., Schroeder, T., Rohde, K., Faithful, J., Masters, B., Dekker, A., Brando, V., Maughan, M. 2010. Dispersal of suspended sediments and nutrients in the Great Barrier Reef lagoon during river-discharge events: conclusions from satellite remote sensing and concurrent flood-plume sampling. Marine and Freshwater Research 61, 651-664.

Brodie, J.E., Kroon, F.J., Schaffelke, B., Wolanski, E.C., Lewis, S.E., Devlin, M.J., Bohnet, I.C., Bainbridge, Z.T., Waterhouse, J., Davis, A.M., 2012. Terrestrial pollutant runoff to the Great Barrier Reef: An update of issues, priorities and management responses. Marine Pollution Bulletin 65, 81-100.

Brodie, J.E., Mitchell, A.W., 2005. Nutrients in Australian tropical rivers: changes with agricultural development and implications for receiving environments. Marine and Freshwater Research 56, 279–302.

Browne, N.K., Smithers, S., Perry C., 2010. Geomorphology and community structure of Middle Reef, central Great Barrier Reef, Australia: an inner-shelf turbid zone reef subject to episodic mortality events. Coral Reefs 29, 683-689.

Browne, N.K., Smithers, S.G., Perry, C.T., 2012. Coral reefs of the turbid inner-shelf of the Great Barrier Reef, Australia: An environmental and geomorphic perspective on their occurrence, composition and growth. Earth Science Reviews 115, 1-20.

Brun, F.G., Hernández, I., Vergara, J.J., Peralta, G., Pérez-Lloréns, J.L., 2002. Assessing the toxicity of ammonium pulses to the survival and growth of *Zostera noltii*. Marine Ecology Progress Series 225, 177–187.

Bruno, J.F., Petes, L.E., Harvell, C.D., Hettinger, A., 2003. Nutrient enrichment can increase the severity of coral diseases. Ecology Letters 6, 1056-1061.

Bruno, J.F., Selig, E.R., Casey, K.S., Page, C.A., Willis, B.L., 2007. Thermal stress and coral cover as drivers of coral disease outbreaks. PLoS Biol 5, e124. doi 10.1371/journal.pbio50124.

Bruno, J.F., Sweatman, H., Precht, W.F., Selig, E.R., Schutte, V.G.W., 2009. Assessing evidence of phase shifts from coral to macroalgal dominance on coral reefs. Ecology 90, 1478-1484.

Bruton, M.N., 1985. The effects of suspensoids on fish. Hydrobiologia 125, 221-241.

Bunn, S.E., Davies, P.M., Kellaway, D.M., Prosser, I.P., 1998. Influence of invasive macrophytes on channel morphology and hydrology in an open tropical lowland stream, and potential control by riparian shading. Freshwater Biology 39, 171-178.

Burkholder, J.M., Mason, K.M., Glasgow, H.B., 1992. Water column nitrate enrichment promotes decline of eelgrass *Zostera marina*: Evidence from seasonal mesocosm experiments. Marine Ecology Progress Series 81, 163–178.

Cabaço, S., Santos, R., Duarte, C.M., 2008. The impact of sediment burial and erosion on seagrasses: A review. Estuarine, Coastal and Shelf Science 79, 354–366.

Calow, P., Sibly, R.M., 1990. A physiological basis of population processes: Ecotoxicological implications. Functional Ecology 4, 283–288.

Campbell, S.J., McKenzie, L.J., 2004. Flood related loss and recovery of intertidal seagrass meadows in southern Queensland, Australia. Estuarine, Coastal and Shelf Science 60, 477-490.

Cantin, N.E., Negri, A.P., Willis, B.L., 2007. Photoinhibition from chronic herbicide exposure reduces reproductive output of reef-building corals. Marine Ecology Progress Series 344, 81-93.

Cantin, N.E., van Oppen, M.J.H., Willis, B.L., Mieog, J.C., Negri, A.P., 2009. Juvenile corals can acquire more carbon from high-performance algal symbionts. Coral Reefs 28, 405-414.

Carilli, J.E., Norris, R.D., Black, B., Walsh, S.M., McField, M., 2010. Century-scale records of coral growth rates indicate that local stressors reduce coral thermal tolerance threshold. Global Change Biology 16, 1247-1257.

Carilli, J.E., Prouty, N.G., Hughen, K.A., Norris, R.D., 2009. Century-scale records of land-based activities recorded in Mesoamerican coral cores. Marine Pollution Bulletin 58, 1835-1842.

Chartrand, K.M., Ralph, P.J., Petrou, K., Rasheed, M.A., 2012. Development of a light-based seagrass management approach for the Gladstone Western basin dredging program, p. 126. Fisheries Queensland, Department of Agriculture Fisheries and Forestry.

Cheal, A.J., MacNeil, M.A., Cripps, E., Emslie, M., Jonker, M., Schaffelke, B., Sweatman, H., 2010. Coral-macroalgal phase shifts or reef resilience: links with diversity and functional roles of herbivorous fishes on the Great Barrier Reef. Coral Reefs 29:, 1005-1015.

Choukroun, S., Ridd, P.V., Brinkman, R., McKinna, L.I.W., 2010. On the surface circulation in the western Coral Sea and residence times in the Great Barrier Reef. Journal of Geophysical Research 115, C06013.

Clough, B.F., Boto, K.G., Attiwill, P.M., 1983. Mangroves and sewerage: a re-evaluation, in Biology and ecology or mangroves, tasks for vegetation science. H.J. Teas, W Junk (eds). Junk Publishers, The Hague, pp. 151-162.

Coles, R., McKenzie, L., De'ath, G., Roelofs, A., Lee Long, W., 2009, Spatial distribution of deepwater seagrass in the inter-reef lagoon of the Great Barrier Reef World Heritage Area. Marine Ecology Progress Series 392, 57–68.

Coles, R., McKenzie, L., Rasheed, M.A., Mellors, J., Taylor, H., Dew, K., Mckenna, S., Sankey, T., Carter, A., Grech, A., 2007. Status and trends of seagrass habitats in the Great Barrier Reef World Heritage Area, p. 122. Unpublished report to the Marine and Tropical Sciences Research Facility. Reef and Rainforest Research Centre Limited.

Collier, C., Waycott, M., 2009. Drivers of change to seagrass distributions and communities on the Great Barrier Reef: Literature review and gaps analysis. Report to the Marine and Tropical Sciences Research Facility. Reef and Rainforest Research Centre Limited.

Collier, C.J., Waycott, M., Giraldo-Ospina, A., 2012a. Responses of four Indo-West Pacific seagrass species to shading. Marine Pollution Bulletin 65, 342-354.

Collier, C.J., Waycott, M., McKenzie, L.J., 2012b. Light thresholds derived from seagrass loss in the coastal zone of the northern Great Barrier Reef, Australia. Ecological Indicators 23, 211-219.

Congdon, R.A., Lukacs, G.P., 1996. Water Quality Aspects of Irrigation Runoff from the Burdekin River Irrigation Area. In: Downstream Effects of Land Use. H. M. Hunter, A.G. Eyles, G. E. Rayment (eds.). Department of Natural Resources, Queensland, Australia. , pp. 73-76.

Connolly, R., Jenkins, G., Loneragan, N., 1999. Seagrass dynamics and fisheries sustainability. In Butler A, Jernakoff P (eds) Seagrasses in Australia: strategic review and development of an R & D plan. CSIRO Publishing, Collingwood, Victoria. pp.25-64

Connolly, S.R., Lopez-Yglesias, M.A., Anthony, K.R.N., 2012. Food availability promotes rapid recovery from thermal stress in a scleractinian coral. Coral Reefs 31, 951-960.

Conolly, N., Kahler, C., Mackay, S., Fry, S., Cameron, R., 2012. Variations in wetland condition across Land Zones in the lower Burdekin. Aquatic weed distribution determined by underlying differences in water and salinity regimes. A report prepared for NQ Dry Tropics NRM. Department of Environment and Heritage Protection, Queensland.

Cooper, T.F., Ridd, P.V., Ulstrup, K.E., Humphrey, C., Slivkoff, M., Fabricius, K.E., 2008. Temporal dynamics in coral bioindicators for water quality on coastal coral reefs of the Great Barrier Reef. Marine and Freshwater Research 59, 703-716.

Cooper, T.F., Uthicke, S., Humphrey, C., Fabricius, K.E., 2007. Gradients in water column nutrients, sediment parameters, irradiance and coral reef development in the Whitsunday Region, central Great Barrier Reef. Estuarine, Coastal and Shelf Science 74, 458-470.

Costanza, R., Bohensky, B., Butler, J., Bohnet, I., Delisle, A., Fabricius, K., Gooch, M., Kubiszewski, I., Lukacs, G., Pert, P., Wolanski, E., 2011. A scenario analysis of climate change and ecosystem services for the Great Barrier Reef. Chapter 16 in Volume 12: Ecological Economics of Estuaries and Coasts, (eds. M. van den Belt and R. Costanza) in the Treatise on Estuarine and Coastal Science (Series eds. E. Wolanski, and D. McLusky), Elsevier.

Cunning, R., Baker, A.C., 2012. Excess algal symbionts increase the susceptibility of reef corals to bleaching. Nature Climate Change doi:10.1038/nclimate1711.

Davis, A.M., Lewis, S.E., Bainbridge, Brodie, J.E., Shannon, E., 2008. Pesticide residues in waterways of the lower Burdekin region: Challenges in ecotoxicological interpretation of monitoring data. Australasian Journal of Ecotoxicology 14, 89-108.

Davis, A.M., Lewis, S.E., Bainbridge, Z.T., Glendenning, L., Turner, R.D.R., Brodie, J.E., 2012. Dynamics of herbicide transport and partitioning under event flow conditions in the lower Burdekin region, Australia. Marine Pollution Bulletin 65, 182-193.

Davis, A.M., Thorburn, P.J., Lewis, S.E., Bainbridge, Z.T., Attard, S.J., Milla, R., Brodie, J.E., in press. Herbicide run-off dynamics of furrow irrigated sugarcane farms and associated drainage systems on the Burdekin River Floodplain, north-eastern Australia. Agriculture, Ecosystems and the Environment doi:10.1016/j.agee.2011.06.019

De'ath, G., Fabricius, K., 2010. Water quality as a regional driver of coral biodiversity and macroalgae on the Great Barrier Reef. Ecological Applications 20, 840-850.

De'ath, G., Fabricius, K.E., Sweatman, H., Puotinen, M., 2012. The 27-year decline of coral cover on the Great Barrier Reef and its causes. Proceedings of the National Academy of Sciences 190, 17995-17999.

De'ath, G., Fabricius, K.E., 2008. Water Quality of the Great Barrier Reef: Distributions, Effects on Reef Biota and Trigger Values for the Protection of Ecosystem Health. Research Publication No. 89. Great Barrier Marine Park Authority, Townsville.

Department of the Premier and Cabinet, 2013. Great Barrier Reef Second Report Card 2010. Reef Water Quality Protection Plan. Australian Government and Queensland Government. Viewed 11/07/2013 http://www.reefplan.qld.gov.au/measuring-success/report-cards/second-report-card/catchment-indicators/wetlands.aspx

Devlin, M., Brodie, J., 2005. Terrestrial discharge into the Great Barrier Reef Lagoon: nutrient behaviour in coastal waters. Marine Pollution Bulletin 51, 9-22.

Devlin, M., Harkness, P., McKinna, L. and Waterhouse, J., 2010. Mapping of risk and exposure of Great Barrier Reef ecosystems to anthropogenic water quality. ACTFR report number 10/12. 2010. James Cook University.

Devlin, M., Schaffelke, B., 2009. Spatial extent of riverine flood plumes and exposure of marine ecosystems in the Tully coastal region, Great Barrier Reef. Marine and Freshwater Research 60, 1109-1122.

Devlin, M.J., McKinna, L.W., Álvarez-Romero, J.G., Petus, C., Abott, B., Harkness, P., Brodie, J., 2012. Mapping the pollutants in surface riverine flood plume waters in the Great Barrier Reef, Australia. Marine Pollution Bulletin 65, 224-235.

Done, T.J., 1982. Patterns in the distribution of coral communities across the central Great Barrier Reef. Coral Reefs 1, 95-107.

Donner, S. D., 2009. Coping with commitment: projected thermal stress on coral reefs under different future scenarios. PLoS ONE 4:e5712.

Dubinsky, Z., Stambler, N., Ben-Zion, M., McClosky, L., Muscatine, L., Falkowski, P., 1990. The effect of external nutrient resources on the optical properties and photosynthetic efficiency of *Stylophora pistillata*. Proceedings of the Royal Society London, Biological Sciences 239, 231-246.

Duke, N.C., 2008. Herbicides implicated as the cause of severe mangrove dieback in the Mackay region, NE Australia - Serious implications for marine plant habitats of the Great Barrier Reef World Heritage Area. Marine Pollution Bulletin 56, 1668-1670.

Duke, N.C.,Wolanski, E., 2001. Muddy coastal waters and depleted mangrove coastlines - depleted seagrass and coral reefs. In: E. Wolanski (ed). Oceanographic processes of coral reefs: physical and biological links in the Great Barrier Reef. CRC Press, Boca Raton, USA. pp. 77-91.

Duke, N.C., Bell, A.M., Pederson, D.K., Roelfsema, C.M., Nash, S.B., 2005. Herbicides implicated as the cause of severe mangrove dieback in the Mackay region, NE Australia: consequences for marine plant habitats of the Great Barrier Reef World Heritage Area. Marine Pollution Bulletin 51, 308-324.

Duke, N.C., Lawn, P., Roelfsema, C.M., Zahmel, K.N., Pederson, D.K., Harris, C., Steggles, N., Tack, C., 2003. Assessing historical change in coastal environments - Port Curtis, Fitzroy River Estuary and Moreton Bay Regions. Historical Coastlines, CRC for Coastal Zone Estuary & Waterways Management, Brisbane, Australia.

Ebert, S.P., 1995. The geomorphological response to sediment discharge from the Herbert River, North Queensland, 1943-1991, Honours Thesis, James Cook University of North Queensland, Townsville.

Ellison, J.C., 1998. Impacts of Sediment Burial on Mangroves. Marine Pollution Bulletin 37, 8-12.

Environment Australia, 2001. A Directory of Important Wetlands in Australia, Third Edition. Environment Australia, Canberra. Commonwealth of Australia. ISBN 0 642 54721 1. http://www.environment.gov.au/water/publications/environmental/wetlands/pubs/directory.pdf

Erftemeijer, P.L.A., Lewis III, R.R.R., 2006. Environmental impacts of dredging on seagrasses: A review. Marine Pollution Bulletin 52, 1553–1572.

Erftemeijer, P.L.A., Reigl, B., Hoeksema, B.W., Todd, P.A., 2012. Environmental impacts of dredging and other sediment disturbances on corals: A review. Marine Pollution Bulletin 64, 1737-1765.

Eslami-Andargoli, L., Dale, P., Sipe, N., Chaseling, J., 2010. Local and landscape effects on spatial patterns of mangrove forest during wetter and drier periods: Moreton Bay, Southeast Queensland, Australia. Estuarine, Coastal and Shelf Science 89, 53-61.

Fabricius, K.E., 2011. Factors determining the resilience of coral reefs to eutrophication: a review and conceptual model. Pages 493-506 *in* Z. Dubinsky and N. Stambler, editors. Coral Reefs: An Ecosystem in Transition, Springer Press.

Fabricius, K., 2006. Effects of irradiance, flow and colony pigmentation on the temperature microenvironment around corals: implications for coral bleaching? Limnology and Oceanography 51, 30-37.

Fabricius, K., Wolanski, E., 2000. Rapid smothering of coral reef organisms by muddy marine snow. Estuarine, Coastal and Shelf Science 50, 115-120.

Fabricius, K.E. 2005. Effects of terrestrial runoff on the ecology of corals and coral reefs: Review and synthesis. Marine Pollution Bulletin 50, 125-146.

Fabricius, K.E., Cooper, T.F., Humphrey, C., Uthicke, S., De'ath, G., Davidson, J., LeGrand, H., Thompson, A., Schaffelke, B., 2012. A bioindicator system for water quality on inshore coral reefs of the Great Barrier Reef. Marine Pollution Bulletin 65, 320-332.

Fabricius, K.E., Cséke, S., Humphrey, C., De'ath, G., 2013b. Does Trophic Status Enhance or Reduce the Thermal Tolerance of Scleractinian Corals? A Review, Experiment and Conceptual Framework. PLoS ONE 8, e54399.

Fabricius, K.E., De'ath, G., Humphrey, C., Zagorskis, I., Schaffelke, B., 2013a. Intra-annual variation in turbidity in response to terrestrial runoff on near-shore coral reefs of the Great Barrier Reef. Estuarine, Coastal and Shelf Science 116, 57-65.

Fabricius, K.E., De'ath, G., McCook, L., Turak, E., Williams, D.McB., 2005. Changes in algal, coral and fish assemblages along water quality gradients on the inshore Great Barrier Reef. Marine Pollution Bulletin 51, 384-396.

Fabricius, K.E., Okaji, K., De'ath, G., 2010. Three lines of evidence to link outbreaks of the crown-of-thorns seastar *Acanthaster planci* to the release of larval food limitation. Coral Reefs 29, 593-605.

Faithful, J., Brodie, J., Bainbridge, Z., Schaffelke, B., Slivkoff, M., Maughan, M., Liessmann, L., Sydes, D., 2008. Water Quality Characteristics of Water Draining Different Land Uses in the Tully/Murray Rivers Region-Edition 2. Report for the Terrain Tully Water Quality Improvement Plan. ACTFR Report No. 08/03. Australian Centre for Tropical Freshwater Research, James Cook University, Townsville. p 85.

Falkowski, P.G., Jokiel, P.L., Kinzie, R.A., 1990. Irradiance and corals. In Dubinsky Z., ed. Ecosystems of the world 25: Coral Reefs. Elsevier, New York, pp 89-107.

Feller, I.C., 1995. Effects of nutrient enrichment on growth and herbivory of dwarf red mangrove (*Rhizophora mangle*). Ecological Monographs 65, 477-505.

Feller, I.C., Lovelock, C.E. and Piou, C., 2009. Growth and nutrient conservation in *Rhizophora* mangle in response to fertilization along latitudinal and tidal gradients. Smithsonian Contributions to Marine Science 38, 345-348.

Feller, I.C., Whigham, D.F., McKee, K.L.,Lovelock, C.E., 2003. Nitrogen limitation of growth and nutrient dynamics in a mangrove forest, Indian River Lagoon, Florida. Oecologia 134, 405-414.

Feller, I.C., Whigham, D.F., McKee, K.L.,O'Neill, J.P., 2002. Nitrogen vs. phosphorus limitation across an ecotonal gradient in a mangrove forest. Biogeochemistry 62, 145-175.

Ferrari, R., Gonzalez-Rivera, M., Mumby, P., 2012. Size matters in competition between corals and macroalgae. Marine Ecology Progress Series 467, 77-88.

Ferrier-Pages, C., Rottier, C., Beraud, E., Levy, O., 2010. Experimental assessment of the feeding effort of three scleractinian coral species during a thermal stress: Effect on the rates of photosynthesis. Journal of Experimental Marine Biology and Ecology 390, 118-124.

Finlayson, B., Silburn, M., 1996. Soil, Nutrient and Pesticide Movements from Different Land Use Practices, and Subsequent Transport by Rivers and Streams. In: Downstream Effects of Land Use. H. M. Hunter, A.G. Eyles, G. E. Rayment (eds.). Department of Natural Resources, Queensland, Australia. pp. 129-40.

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, 677-685.

Flores, F., Hoogenboom, M.O., Smith, L.D., Cooper, T.F., Abrego, D., Negri ,A.P., 2012. Chronic exposure of corals to fine sediments: lethal and sub-lethal impacts. PLoS ONE 7, e37795.

Foster, N.L., Box, S.J., Mumby, P.J., 2008. Competitive effects of macroalgae on the fecundity of the reef-building coral *Montastrea annularis*. Marine Ecology Progress Series 367, 143-152.

Fourqurean, J.W., Willsie, A., Rose, C.D., Rutten, L.M., 2001. Spatial and temporal pattern in seagrass community composition and productivity in south Florida. Marine Biology 138, 341-354.

Furnas, M. Mitchell, A. Skuza, M. 1997. Shelf-scale nitrogen and phosphorus budgets for the central Great Barrier Reef (16-19°S). Proceedings of the 8th International Coral Reef Symposium 1, 809-814.

Furnas, M., Mitchell, A., Skuza, M., Brodie, J.E., 2005. In the other 90%: phytoplankton responses to enhanced nutrient availability in the Great Barrier Reef lagoon. Marine Pollution Bulletin 51, 253-265.

Furnas, M., Alongi D., McKinnon, D., Trott, L., Skuza, M., 2011. Regional-scale nitrogen and phosphorus budgets for the northern (14°S) and central (17°S) Great Barrier Reef shelf ecosystem. Continental Shelf Research 31, 1967-1990.

Furnas, M.J., 1989. Cyclonic disturbance and a phytoplankton bloom in a tropical shelf ecosystem, in: Okaichi, T. Anderson, D.M. Nemoto, T. (Eds.), Red Tides: Biology, Environmental Science and Toxicology. Proceedings of the First International Symposium on Red Tides held November 10-14, 1987, in Takamatsu, Kagawa Prefecture, Japan. Elsevier Science Publishing Co. Inc. pp. 273-276.

Furnas, M.J., Mitchell, A.W., Skuza, M., 1995. Nitrogen and phosphorus budgets for the central Great Barrier Reef shelf. Research Publication No. 36, Great Barrier Reef Marine Park Authority, Townsville.

Furnas, M., Brinkman, R., Fabricius, K., Tonin, H., Schaffelke, B., 2013. Chapter 1: Linkages between river runoff, phytoplankton blooms and primary outbreaks of crown-of-thorns starfish in the Northern GBR. In: *Assessment of the* *relative risk of water quality to ecosystems of the Great Barrier Reef: Supporting Studies*. A report to the Department of the Environment and Heritage Protection, Queensland Government, Brisbane. TropWATER Report 13/30, Townsville, Australia.

Gao, Y., Fang, J., Zhang, J., Ren, L., Mao, Y., Li B., Zhang, M., Liu, D., Du, M., 2011. The impact of the herbicide atrazine on growth and photosynthesis of seagrass, *Zostera marina*, seedlings. Marine Pollution Bulletin 62, 1628-1631.

Gleeson, S.K., 1993. Optimization of tissue nitrogen and root-shoot allocation. Annals of Botany 71, 23-31.

Goreau, T., McClanahan, T., Hayes, R., Strong, A., 2000. Conservation of coral reefs after the 1998 global bleaching event. Conservation Biology 14, 5-15.

Great Barrier Reef Marine Park Authority, 2012. Informing the outlook for Great Barrier Reef coastal ecosystems, Great Barrier Reef Marine Park Authority, Townsville.

Great Barrier Reef Marine Park Authority, 2009. Water Quality Guidelines for the Great Barrier Reef Marine Park. Great Barrier Reef Marine Park Authority, Townsville, 99p.

Grech, A., Coles, R., Marsh, H., 2011. A broad-scale assessment of the risk to coastal seagrasses from cumulative threats. Marine Policy 35, 560-567.

Grech, A., Coles, R.G., 2010. An ecosystem-scale predictive model of coastal seagrass distribution. Aquatic conservation: Marine and Freshwater Ecosystems 20, 437-444.

Grottoli, A.G., Rodrigues, L.J., Juarez, C., 2004. Lipids and stable carbon isotopes in two species of Hawaiian corals, *Porites compressa* and *Montipora verrucosa*, following a bleaching event. Marine Biology 145, 621-631.

Grottoli, A.G., Rodrigues, L.J., Palardy, J.E., 2006. Heterotrophic plasticity and resilience in bleached corals. Nature 440, 1186-1189.

Haapkylä, J., Unsworth, R., Flavell, M., Bourne, D., Schaffelke, B., Willis, B.L., 2011. Seasonal rainfall and runoff promote coral disease on an inshore reef of the Great Barrier Reef, Australia. PLoS One. 6: e16893. doi:10.1371/journal.pone.0016893.

Hancock, G.J., Webster, I.T., Stieglitz, T.C., 2006. Horizontal mixing of Great Barrier Reef waters: Offshore diffusivity determined from radium isotope distribution. Journal of Geophysical Research 111, C12019. doi:10.1029/2006JJC003608.

Harrington, L., Fabricius, K., Eaglesham, G., Negri, A., 2005. Synergistic effects of diuron and sedimentation on photosynthesis and survival of crustose coralline algae, Marine Pollution Bulletin 51, 415-427.

Harris, G.P., 2001. Biogeochemistry of nitrogen and phosphorus in Australian catchments, rivers and estuaries: effects of land use and flow regulation and comparisons with global patterns. Marine and Freshwater Research 52, 139-149.

Hauri, C., Fabricius, K., Schaffelke, B., Humphrey, C., 2010. Chemical and physical environmental conditions underneath mat- and canopy-forming macroalgae, and the effects on understory corals. PLoS ONE 5(9): e12685. doi:10.1371/journal.pone.0012685.

Haynes, D., Müller, J., Carter, S., 2000. Pesticide and herbicide residues in sediments and seagrasses from the Great Barrier Reef World Heritage Area and Queensland Coast. Marine Pollution Bulletin 41, 279-287.

Haynes, D., Ralph, P., Prange, J., Dennison, W.C., 2000. The Impact of the Herbicide Diuron on Photosynthesis in Three Species of Tropical Seagrass. Marine Pollution Bulletin 41, 288-293.

Hermens, J., Busser, F., Leeuwangh, P., Musch, A., 1985a. Quantitative structure activity relationships and mixture toxicity of organic chemicals in *Photobacterium phosphoreum*: the Microtox test. Ecotoxicology and Environmental Safety 9, 17-25.

Hermens, J., Canton, H., Janssen, P., De Jong, R., 1984. Quantitative structure-activity relationships and toxicity studies of mixtures of chemicals with anaesthetic potency: acute lethal and sublethal toxicity to *Daphnia magna*. Aquatic Toxicology 5, 143-154.

Hermens, J., Leeuwangh, P., Musch, A., 1985b. Joint toxicity of mixtures of groups of organic aquatic pollutants to the guppy (*Poecilia reticulata*). Ecotoxicology and Environmental Safety 9, 321-326.

Herzfeld, M., Waring, J., Parslow, J., Margvelashvili, N., Sakov, P., Andrewartha, J., 2006. SHOC: Sparse Hydrodynamic Ocean Code Science manual. CSIRO internal document. 121 p.

Hoegh-Guldberg, O., Mumby, P.J., Hooten, A.J., 2007. Coral reefs under rapid climate change and ocean acidification. Science 318, 1737-1742.

Hoogenboom, M. O., S. R. Connolly, and K. R. N. Anthony. 2009. Effects of photoacclimation on the light niche of corals: a process-based approach. Marine Biology 156, 2493–2503.

Hoogenboom, M., Campbell, D., Beraud, E., DeZeeuw, K., Ferrier-Pages, C., 2012. Effects of light, food availability and temperature stress on the function of photosystem II and photosystem I of coral symbionts. PLoS ONE 7:e30167. doi:30110.31371/journal.pone.0030167

Hopley, D. 1974. Coastal changes produced by tropical cyclone Althea in Queensland, December 1971. Australian Geographer 7, 445-456.

Houlbrèque, F., Ferrier-Pages, C., 2009. Heterotrophy in tropical scleractinian corals. Biological Review 84, 1-17

Houston, W.A. 1999. Severe hail damage to mangroves at Port Curtis, Australia. Mangroves and Sea Marshes 3, 29-40.

Howard, G.W., and Harley, K.L.S., 1998. How do floating aquatic weeds affect wetland conservation and development? How can these effects be minimised? Wetlands Ecology and Management 5, 215-225.

Hughes, T.P., Bellwood, D.R., Baird, A.H., Brodie, J., Bruno, J.F., Pandolfi, J.M., 2011. Shifting base-lines, declining coral cover, and the erosion of reef resilience: comment on Sweatman *et al.,* 2011. Coral Reefs 30, 653-660

Hughes, T.P., Rodrigues, M.J., Bellwood, D.R., Ceccarelli, D., Hoegh-Guldberg, O., McCook, L., Moltschaniwskyj, N., Pratchett, M.S., Steneck, R.S., Willis B., 2007. Phase Shifts, Herbivory, and the Resilience of Coral Reefs to Climate Change. Current Biology 17, 360-365.

Jimenez, I.M., Kuhl, M., Larkum, A.W.D., Ralph, P.J., 2008. Heat budget and thermal microenvironment of shallow-water corals: Do massive corals get warmer than branching corals? Limnology and Oceanography 53, 1548-1561.

Johannes, R.E., Wiebe, W.J., Crossland, C.J., Rimmer, D.W., Smith, S.V., 1983. Latitudinal limits of coral reef growth. Marine Ecology Progress Series 11, 105:111.

Johnson, A.K.L., Ebert, S.P., Murray, A.E., 1999. Distribution of coastal freshwater wetlands and riparian forests in the Herbert River catchment and implications for management of catchments adjacent the Great Barrier Reef Marine Park. Environmental Conservation 26,229–235

Jones, R.J., Kerswell, A.P., 2003. Phytotoxicology of photosystem II., PSII. herbicides to coral. Marine Ecology Progress Series 251, 153-167.

Joo, M., Raymond, M., McNeil, V., Huggins, R., Turner, R., Choy, S., 2012. Estimates of sediment and nutrient loads in 10 major catchments draining to the Great Barrier Reef during 2006-2009. Marine Pollution Bulletin 65, 320-332.

Jupiter, S.D., Potts, D.C., Phinn, S.R. and Duke, N.C., 2007. Natural and anthropogenic changes to mangrove distributions in the Pioneer River Estuary (QLD, Australia). Wetlands Ecology and Management 15, 51-62.

Kennedy, K., Devlin, M., Bentley, C., Lee-Chue, K., Paxman, C., Carter, S., Lewis, S., Brodie, J., Guy, E., Vardy, S., Martin, K., Jones, A., Packett, R., Mueller, J., 2012b The influence of a season of extreme wet weather events on exposure of the World Heritage Area Great Barrier Reef to pesticides. Marine Pollution Bulletin 64, 1495–1507

Kennedy, K., Schroeder, T., Shaw, M., Haynes, D., Lewis, S., Bentley, C., Paxman, C., Carter, S., Brando, VE., Bartkow, M., Hearn, L., Mueller, JF., 2012a. Long term monitoring of photosystem II herbicides – Correlation with remotely sensed freshwater extent to monitor changes in the quality of water entering the Great Barrier Reef, Australia. Marine Pollution Bulletin 65, 292-305

King, B., McAllister, F., Done, T., 2000. Modelling the Impact of the Burdekin, Herbert, Tully and Johnstone River Plumes on the Central Great Barrier Reef. Final report to the CRC Reef Research Centre from Asia-Pacific Applied Science Associates and the Australian Institute of Marine Science, Townsville.

King, B., Spagnol, S., Wolanski, E. & Done, T. 1997, 'Modeling the mighty Burdekin River in flood', Fifth International Conference on Coastal and Estuarine Modeling, Washington D.C., 22–24 October 1997.

King, J., Alexander, F., Brodie, J., 2012. Regulation of pesticides in Australia: The Great Barrier Reef as a case study for evaluating effectiveness. Agriculture, Ecosystems and Environment, in press.

Kirk, J.T.O., 2011. Light and photosynthesis in aquatic ecosystems. Cambridge University Press.

Kirkwood, A. and Dowling, R. 2002. Investigation of Mangrove Dieback: Pioneer River Estuary, Mackay. Queensland Herbarium, Environmental Protection Agency, Brisbane, Australia.

Kleypas, J. A. 1996. Coral reef development under naturally turbid conditions: fringing reefs near Broad Sound, Australia. Coral Reefs **15**, 153-167.

Kleypas, J.A., Buddemeier, R.W., Gattuso, J.P., 2001. The future of coral reefs in an age of global change. International Journal of Earth Sciences 90, 426:437.

Knutson, T.R., McBride, J.L., Chan, J., Emanuel, K., Holland, G., Landsea, C., Held, I., Kossin, J.P., Sirvastava, A.K., Sugi, M., 2010. Tropical cyclones and climate change. Nature Geoscience 3, 157:163.

Könemann, H., 1980, Structure-activity relationships and additivity in fish toxicities of environmental pollutants. Ecotoxicology and Environmental Safety 4, 15-421.

Kortenkamp, A., Backhaus, T., Faust, M., 2009. State of the art report on mixture toxicity. Final report. Report to the Commission of the European Union (Directorate General for the Environment). EC Report no. 070307/2007/485103/ETU/D.1. Available from: http://ec.europa.eu/environment/chemicals/pdf/report_Mixture%20toxicity.pdf.

Kroon, F.J., Kuhnert, P.M., Henderson, B.L., Wilkinson, S.N., Kinsey-Henderson, A., Abbott, B., Brodie, J.E., Turner, R.D.R., 2012. River loads of suspended solids, nitrogen, phosphorus and herbicides delivered to the Great Barrier Reef lagoon. Marine Pollution Bulletin 65, 167-181.

Kroon, F., Turner, R.D.R., Smith, R., Warne, M.St.J., Hunter, H., Bartley, R., Wilkinson, S., Lewis, S., Waters, D., Carroll, C., 2013. Chapter 4: Sources of sediment, nutrients, pesticides and other pollutants in the Great Barrier Reef catchment. Reef Water Quality Scientific Consensus Statement 2013. Department of the Premier and Cabinet, Queensland Government, Brisbane.

Lambrechts, J. Humphrey, C. McKinna, L. Gourge, O. Fabricius, K. Mehta, A. Lewis, S. Wolanski, E. 2010. The importance of wave-induced bed fluidisation in the fine sediment budget of Cleveland Bay, Great Barrier Reef. Estuarine, Coastal and Shelf Science 89, 154-162.

Larcombe, P. Ridd, P.V. Prytz, A. Wilson, B. 1995. Factors controlling suspended sediment on inner-shelf coral reefs, Townsville, Australia. Coral Reefs 14, 163-171.

Larcombe, P., Costen, A., Woolfe, K., 2001. The hydrodynamic and sedimentary setting of nearshore coral reefs, central Great Barrier Reef, Australia: Paluma Shoals, a case study. Sedimentology 48, 811-835.

Larcombe, P., Ridd, P.V., Wilson, B., Prytz, A., 1995. Factors controlling suspended sediment on inner-shelf coral reefs, Townsville, Australia. Coral Reefs 14, 163-171

Lee, J.H., Landrum, P.F., 2006a. Application of multi-component damage assessment model (MDAM) for the toxicity of metabolized PAH in *Hyalella azteca*. Environmental Science and Technology 40, 1350–1357.

Lee, J.H., Landrum, P.F., 2006b. Development of a multi-component damage assessment model (MDAM) for time-dependent mixture toxicity with toxicokinetic interactions. Environmental Science and Technology 40, 1341–1349.

Lewis, S.E., Brodie, J.E., Bainbridge, Z.T., Rohde, K.W., Davis, A.M., Masters, B.L., Maughan, M., Devlin, M.J., Mueller, J.F., Schaffelke, B., 2009. Herbicides: A new threat to the Great Barrier Reef. Environmental Pollution 157, 2470-2484.

Lewis, S.E. Smith, R. Brodie, J.E. Bainbridge, Z.T. Davis, A.M. Turner, R. (2011). Using monitoring data to model herbicides exported to the Great Barrier Reef, Australia. In Chan, F., Marinova, D. and Anderssen, R.S. (eds) MODSIM2011, 19th International Congress on Modelling and Simulation. Modelling and Simulation Society of Australia and New Zealand, December 2011, pp. 2051-2056. ISBN: 978-0-9872143-1-7. www.mssanz.org.au/modsim2011/E5/lewis.pdf

Lewis, S.E., Brodie, J.E., McCulloch, M.T., Mallela, J., Jupiter, S.D., Stuart Williams, H., Lough, J.M., Matson, E.G., 2012b. An assessment of an environmental gradient using coral geochemical records, Whitsunday Islands, Great Barrier Reef, Australia. Marine Pollution Bulletin 65, 306-319.

Lewis, S.E., Schaffelke, B., Shaw M., Bainbridge, Z.T., Rohde, K.W., Kennedy, K., Davis, A.M., Masters, B.L., Devlin, M.J., Mueller, J.F., Brodie, J.E., 2012a. Assessing the additive risks of PSII herbicide exposure to the Great Barrier Reef. Marine Pollution Bulletin 65, 280-291.

Lewis, S., Smith, R., O'Brien, D., Warne, M. St. J., Negri, A., Petus, C., da Silva, E., Zeh, D. Turner, R.D.R., Davis, A., Mueller, J., Brodie, J., 2013. Chapter 4: Assessing the risk of additive pesticide exposure in Great Barrier Reef ecosystems. In: *Assessment of the relative risk of water quality to ecosystems of the Great Barrier Reef: Supporting Studies*. A report to the Department of the Environment and Heritage Protection, Queensland Government, Brisbane. TropWATER Report 13/30, Townsville, Australia.Lough, J., 2011. Great Barrier Reef coral luminescence reveals rainfall variability over north-eastern Australia sine the 17th century. Paleoceanography 26, PA2201, doi:10.1029/2010PA002050.

Lovelock, C.E. and Feller, I.C. 2003. Photosynthetic performance and resource utilization of two mangrove species coexisting in a hypersaline scrub forest. Oecologia 134, 455-462.

Lovelock, C.E., Ball, M.C., Choat, B., Engelbrecht, B.M., Holbrook, N.M. and Feller, I.C. 2006a. Linking physiological processes with mangrove forest structure: phosphorus deficiency limits canopy development, hydraulic conductivity and photosynthetic carbon gain in dwarf *Rhizophora* mangle. Plant, Cell and Environment 29, 793-802.

Lovelock, C.E., Ball, M.C., Feller, I.C., Engelbrecht, B.M.J., Ewe, M.L. 2006b. Plant function in nitrogen and phosphorus limited mangrove ecosystems. New Phytolologist 172, 514–522.

Lovelock, C.E., Ball, M.C., Martin, K.C., Feller, I. and Thompson, R. 2009. Nutrient enrichment increases mortality of mangroves. PLoS ONE 4:e5600.

Lovelock, C.E., Bennion, V., Grinham, A. and Cahoon, D.R. 2011a, The Role of Surface and Subsurface Processes in Keeping Pace with Sea Level Rise in Intertidal Wetlands of Moreton Bay, Queensland, Australia. Ecosystems (14), 745-757.

Lovelock, C.E., Feller, I.C., Adame, M.F., Reef, R., Penrose, H.M., Wei, L., and Ball, M.C. 2011b. Intense storms and the delivery of materials that relieve nutrient limitations in mangroves of an arid zone estuary. Functional Plant Biology 38, 514–522

Lovelock, C.E., Feller, I.C., Ball, M.C., Ellis, J. and Sorrell, B. 2007. Testing the growth rate vs. geochemical hypothesis for latitudinal variation in plant nutrients. Ecology Letters 10, 1154-1163.

Lovelock, C.E., Feller, I.C., McKee, K.L., Engelbrecht, B.M. and Ball, M.C. 2004. The effect of nutrient enrichment on growth, photosynthesis and hydraulic conductance of dwarf mangroves in Panama. Functional Ecology 18, 25-33.

Lovelock, C.E., Skilleter, G. and Saintilan, N. 2012. Tidal Wetlands. In: Marine Climate Change Impacts and Adaptation Report Card for Australia 2012. E.S. Poloczanska, A.J. Hobday and A.J. Richardson (eds). Available at: www.oceanclimatechange.org.au.

Luick, J.L., Mason, L., Hardy, T., Furnas, M.J., 2007. Circulation in the Great Barrier Reef Lagoon using numerical tracers and in situ data. Continental Shelf Research 27, 757-778.

MacKinnon, M., Herbert, B., 1996. Temperature, dissolved oxygen and stratification in a tropical reservoir, Lake Tinaroo, North Queensland, Australia. Marine and Freshwater Research 47(7): 937-949. doi:10.1071/MF9960937

Magnusson, M., Heimann, K., Negri, A.P., 2008. Comparative effects of herbicides on photosynthesis and growth of tropical estuarine microalgae. Marine Pollution Bulletin 56, 1545-1552.

Magnusson, M., Heimann, K., Quayle, P., Negri, A.P., 2010. Additive toxicity of herbicide mixtures and comparative sensitivity of tropical benthic microalgae. Marine Pollution Bulletin 60, 1978-1987.

Magnusson, M., Heimann, K., Ridd, M., Negri, A.P., 2012. Chronic herbicide exposures affect the sensitivity and community structure of tropical benthic microalgae. Marine Pollution Bulletin 65, 363-372.

Markey, K.L., Baird, A.H., Humphrey, C., Negri, A.P., 2007. Insecticides and a fungicide affect multiple coral life stages. Marine Ecology Progress Series 330, 127-137.

Marsh, H., O'Shea, T.J., Reynolds III, J.E., 2011. Ecology and conservation of the Sirenia: dugongs and manatees. Cambridge University Press, New York. 536pp.

Martin, K.C., 2007. Interactive effects of salinity and nutrients on mangrove physiology: implications for mangrove forest structure and function. Ph.D. Thesis, ANU, Canberra .

Martin, K.C., Bruhn, D., Lovelock, C.E., Feller, I.C., Evans, J.R. and Ball, M.C. 2010. Nitrogen fertilization enhances water-use efficiency in a saline environment. Plant, Cell and Environment 33, 344-357.

McCook, L.J., Jompa, J., Diaz-Pulido, G., 2001. Competition between corals and algae on coral reefs: a review of evidence and mechanisms. Coral Reefs 19, 400-417.

McCulloch, M.T. Fallon, S. Wyndham, T. Hendy, E. Lough, J. Barnes, D. 2003. Coral record of increased sediment flux to the inner Great Barrier Reef since European settlement. Nature 421, 727–730.

McKee, K.L., 1996. Growth and physiological responses of neotropical mangrove seedlings to root zone hypoxia. Tree Physiology 16, 883-889.

McKee, K.L., Cahoon, D.R. and Feller, I.C., 2007. Caribbean mangroves adjust to rising sea level through biotic controls on change in soil elevation. Global Ecology and Biogeography 16, 545-556.

McKenzie, L., Collier, C., Waycott, M., Unsworth, R., Yoshida, R., Smith, N., 2012b. Monitoring inshore seagrasses of the Great Barrier Reef and responses to water quality. Proceedings of the 12th International Coral Reef Symposium, Cairns, Australia, 9-13 July 2012. Mini-Symposium 15b – Seagrasses and seagrass ecosystems. Available at: http://www.icrs2012.com/proceedings/manuscripts/ICRS2012_15B_4.pdfhttp://www.icrs2012.com/proceedings/manuscripts/ICRS2012_15B_4.pdf.

McKenzie, L.J., Collier, C., Waycott, M., 2012a. Reef Rescue Marine Monitoring Program - Inshore Seagrass, Annual Report for the sampling period 1st July 2010 – 31st May 2011. Fisheries Queensland, Cairns. 230pp.

McKenzie, L.J., Unsworth, R.K.F., Waycott, M., 2010. Reef Rescue Marine Monitoring Program: Intertidal Seagrass, Annual Report for the sampling period 1st September 2009 – 31st May 2010. Fisheries Queensland, Cairns (136pp).

McKinnon, A. D., M. Logan, S. Castine and S. Duggan (accepted). Pelagic metabolism in the waters of the Great Barrier Reef. Limnology & Oceanography

McMahon, K.M., Bengston-Nash, S., Eaglesham, G.K., Mueller, J., Duke, N.C., Winderlich, S., 2005. Herbicide contamination and the potential impact to seagrass meadows in Hervey Bay, Queensland, Australia. Marine Pollution Bulletin 51, 325-334

Mellors, J., Waycott, M., Marsh, H., 2005. Variation in biogeochemical parameters across intertidal seagrass meadows in the central Great Barrier Reef region. Marine Pollution Bulletin 51, 335-342

Morrow, K.M., Ritson-Williams, R., Ross, C., Liles, M.R., Paul, V.J., 2012. Macroalgal extracts induce bacterial assemblage shifts and sublethal tissue stress in Caribbean corals. PLoS ONE 7(9):e44859.doi:10.1371/journal.pone.0044859.

Moss, R.H., Edmonds, J.A., Hibbard, K.A., Manning, M.R., Rose, S.K., van Vuuren, D.F., Carter, T.R., Emori, S., Kainuma, M., Tom Kram, T., Gerald A. Meehl, G., John F. B. Mitchell, J.F.B., Nakicenovic, N., Riahi, K., Steven J. Smith, S., Stouffer, R.J., Thomson, A.M., Weyant, J.P., Wilbanks, T.J., 2010. The next generation of scenarios for climate change research and assessment. Nature 463, 747-756.Muscatine, L., 1990. The role of symbiotic algae in carbon and energy flux in reef corals, pp. 75-87, In Dubinsky Z. (ed.), Ecosystems of the world: Coral Reefs. Elsevier.

Naidoo, G. 2009. Differential effects of nitrogen and phosphorus enrichment on growth of dwarf Avicennia marina mangroves. Aquatic Botany 90,184-190.

Negri, A.P., Flores, F., Röthig, T., Uthicke, S., 2011. Herbicides increase the vulnerability of corals to rising sea surface temperature. Limnology and Oceanography 56, 471-485.

Negri, A.P., Vollhardt, C., Humphrey, C., Heyward, A., Jones, R., Eaglesham, G., Fabricius, K., 2005. Effects of the herbicide diuron on the early life history stages of coral. Marine Pollution Bulletin 51, 370-383.

Negri, A.P., Webster, N.S., Hill, R.T., Heyward, A.J., 2002. Metamorphosis of broadcast spawning corals in response to bacteria isolated from crustose algae. Marine Ecology Progress Series 223, 121-131.

Nobes, K., Uthicke, S., Henderson, R., 2008. Is light the limiting factor for the distribution of benthic symbiont bearing Foraminifera on the Great Barrier Reef? Journal of Experimental Marine Biology and Ecology 363, 48-57

Nordemar, I., Nystroem, M., Dizon, R., 2003. Effects of elevated seawater temperature and nitrate enrichment on the branching coral *Porites cylindrica* in the absence of particulate food. Marine Biology 142, 669-677.

Orpin, A.R., Ridd, P.V., 2012. Exposure of inshore corals to suspended sediments due to wave-re-suspension and river plumes in the central Great Barrier Reef: A reappraisal. Continental Shelf Science 47(15), 55-67.

Osborne, K., Dolman, A.M., Burgess, S.C., Johns, K.A., 2011. Disturbance and the Dynamics of Coral Cover on the Great Barrier Reef., 1995–2009.. PLoS ONE 6:e17516.

Packett, R., Dougall, C., Rohde, K., Noble, R. 2009. Agricultural lands are hot-spots for annual runoff polluting the southern Great Barrier Reef lagoon. Marine Pollution Bulletin 58, 976-986.

Paling, E.I., Kobryn, H.T. and Humphreys, C. 2008. Assessing the extent of mangrove change caused by Cyclone Vance in the eastern Exmouth Gulf, northwestern Australia. Estuarine, Coastal and Shelf Science 77, 603-613.

Palmer, S.E., Perry, C.T., Smithers, S.G., and Gulliver, P. 2010. Internal structure and accretionary history of a nearshore, turbid-zone coral reef: Paluma Shoals, central Great Barrier Reef, Australia: Marine Geology, v. 276, p. 14-29.

Perna, C.N., Cappo, M., Pusey, B.J., Burrows, D.W., Pearson, R.G., 2012. Removal of aquatic weeds greatly enhances fish community richness and diversity: an example from the Burdekin River floodplain, tropical Australia. River Research and Applications 28(8), 1093-1104.

Perry, C.T., Smithers, S.G. 2006. Taphonomic signatures of turbid-zone reef development: Examples from Paluma Shoals and Lugger Shoal, inshore central Great Barrier Reef, Australia. Palaeogeography, Palaeoclimatology, Palaeoecology 242, 1-20.

Perry, C.T., Smithers, S.G. 2010. Evidence for the episodic “turn on” and “turn off” of turbid-zone coral reefs during the late Holocene sea-level highstand. Geology 38, 119-122.

Perry, C.T., Smithers, S.G. 2011. Cycles of coral reef ‘turn-on’, rapid growth and ‘turn-off’ over the past 8500 years: a context for understanding modern ecological states and trajectories. Global Change Biology 17. 76-86.

Perry, C.T., Smithers, S.G, Perna, C., O’Connor, R., and Cook, B. (2012). Hydroecology of the lower Burdekin River alluvial aquifer and associated groundwater dependent ecosystems. Department of Environment and Resource Management, Queensland, Brisbane.

Perry, C.T., Smithers, S.G., Gulliver, P., Browne, N.K., 2012. Evidence of very rapid reef accretion and reef growth under high turbidity and terrigenous sedimentation. Geology 40, 719-722.

Perry, C.T., Smithers, S.G., Johnson, K.G. 2009. Long-term coral community records from Lugger Shoal on the terrigenous inner shelf of the central Great Barrier Reef, Australia. Coral Reefs 28, 941-948.

Perry, C.T., Smithers, S.G., Roche, R.C., Wassenburg, J., 2011, Recurrent patterns of coral community and sediment facies development through successive phases of Holocene inner-shelf reef growth and decline: Marine Geology, v. 289, p. 60-71.

Perry, C., Spencer, T., Kench, P. 2008. Carbonate budgets and reef production states: a geomorphic perspective on the ecological phase-shift concept. Coral Reefs 27, 853-866.

Preen, A.R., Lee Long, W.J., Coles R.G., 1995. Flood and cyclone related loss, and partial recovery, of more than 1000 km2 of seagrass in Hervey Bay, Queensland, Australia. Aquatic Botany 52, 3-17.

Preen, A.R., Marsh, H., 1995. Response of dugongs to large-scale loss of seagrass from Hervey Bay, Queensland, Australia. Wildlife Research 22, 507-519.

Pringle, A.W., 1989. The history of dredging in Cleveland Bay, Queensland and its effect on sediment movement and on the growth of mangroves, corals and seagrass, Vol. Great Barrier Reef Marine Park Authority, Townsville

Queensland Wetlands Program, no date. Healeys Lagoon. Available at: http://wetlandinfo.ehp.qld.gov.au/resources/static/pdf/resources/tools/gbr-case-studies/epa09_026_web_case_studies_healeys_v7.pdf

Radke, L.C., Ford, P.W., Webster, I.T., Atkinson, I., Douglas, G., Oubelkheir, K., Li, J., Robson, B., Brooke, B., 2010. Biogeochemical zones within a macrotidal, dry-tropical fluvial-marine transition area: a dry-season perspective. Aquatic Geochemistry 16, 1–29.

Ralph, P., Tomasko, D.A., Moore, K.A., Seddon, S., Macinnis-Ng, C.M.O., 2006. Human impacts on seagrasses: Eutrophication, sedimentation, and contamination. In: A. W. D. Larkum, R. J. Orth, C. Duarte (eds). Seagrasses: Biology, ecology and conservation. Dordrecht, The Netherlands.

Reddy, K.R. and D'Angelo, E.M.D. 1997. Biogeochemical indicators to evaluate pollutant removal efficiency in constructed wetlands. Water, Science and Technology 35, 1-10.

Reef, R., Feller, I.C. and Lovelock, C.E., 2010. Nutrition of mangroves. Tree Physiology 30, 1148-1160.

Reymond, C.E., Uthicke, S., Pandolfi, J.M., 2011. Inhibited growth in the photosymbiont-bearing foraminifer *Marginopora vertebralis* from the nearshore Great Barrier Reef, Australia. Marine Ecology Progress Series 435: 97-109

Reymond, C.E., Uthicke, S., Pandolfi, J.M., 2012. Tropical Foraminifera as indicators of water quality and temperature. Proceedings of the 12th International Coral Reef Symposium, Cairns, Australia, 9-13 July 2012, 21B Enhancing coral reef resilience through management of water quality, D. Yellowlees and T. P. Hughes (eds.), James Cook University, Townsville, Queensland 4811, Australia

Richmond, R.H., Wolanski, E., 2011. Coral research: past efforts and future horizons, in Z. Dubinsky and N. Stambler (Eds.), Coral Reefs: An Ecosystem in Transition. Springer Science, 552 pp. ISBN: 978-94-007-0113-7, pp. 3-10.

Robson, B.J., Hamilton, D.P., Webster, I.T., Chan, T. 2008. Ten steps applied to development and evaluation of process-based biogeochemical models of estuaries. Environmental Modelling and Software 23, 369-384.

Roche, R.C., Perry, C.T., Johnson, K.G., Sultana, K., Smithers, S.G., Thompson, A.A., 2011. Mid-Holocene coral community data as baselines for understanding contemporary reef ecological states. Palaeogeogr. Palaeoclimatol. Paleaoecol.299, 159–167.

Roff, G., Clark, T.R., Reymond, C.E., Zhao, J.-X., Feng, Y., McCook, L.J., Done, T.J., Pandolfi, J.M., 2013. Palaeoecological evidence of a historical collapse of corals at Pelorus Island, inshore Great Barrier Reef, following European settlement. Proceedings of the Royal Society B: Biological Sciences 280, 20122100.

Rogers, C.S., 1990. Responses of coral reefs and reef organisms to sedimentation. Marine Ecology Progress Series 62, 185-202.

Rohde, K., Masters, B., Brodie, J., Faithful, J., Noble, R., Carroll, C., Shaw, C., Dekker, A., Brando, V. 2006. Fresh and marine water quality in the Mackay Whitsunday region, 2004–2005. Mackay Whitsunday Natural Resource Management Group, Mackay.

Romero, J., Lee, K., Perez, M., Mateo, MA. and Alcoverro, T. 2006. Nutrient Dynamics in Seagrass Ecosystems. Chapter 9 In Larkum, AWD, Orth, RJ and Duarte, CM, (Eds) Seagrasses: Biology, Ecology and Conservation. Springer, Netherlands. pp. 227-254

Rose, R.M., Warne, M. St.J., Lim, R.P., 2000. Life history responses of the cladoceran *Ceriodaphnia* cf. *dubia* to variation in food concentration. Hydrobiologia 427, 59-64.

Ryan, D.A., Brooke, B.P., Bostock, H.C., Radke, L.C., Siwabessy, P.J.W., Margvelashvili, N., and Skene, D. 2007. Bedload sediment transport dynamics in a macro tidal embayment, and implications for export to the southern Great Barrier Reef shelf. Marine Geology 240, 197-215.

Ryan, P. 1991. Environmental effects of sediment on New Zealand streams:A review. New Zealand Journal of Marine and Freshwater Research 25, 207-221.

Santini, N.S., Schmitz, N., Bennion, V. and Lovelock, C.E. 2012. The anatomical basis of the link between density and mechanical strength in mangrove branches. Functional Plant Biology. doi.org/10.1071/FP12204.

Santos, I. R., R. N. Glud, D. Maher, D. Erler, and B. D. Eyre. 2011. Diel coral reef acidification driven by porewater advection in permeable carbonate sands, Heron Island, Great Barrier Reef. Geophysical Research Letters 38, L03604. doi:03610.01029/02010GL046053.

Schaffelke, B. 1999. Short-term nutrient pulses as tools to assess responses of coral reef macroalgae to enhanced nutrient availability. Marine Ecology Progress Series 182, 305-310.

Schaffelke, B., Carleton, J., Costello, P., Davidson, J., Doyle, J., Furnas, M., Gunn, K., Skuza, M., Wright, M., Zagorskis, I., 2012b. Reef Rescue Marine Monitoring Program. Final Report of AIMS Activities 2010/11– Inshore Water Quality Monitoring. Report for the Great Barrier Reef Marine Park Authority. Australian Institute of Marine Science, Townsville. 94 p.

Schaffelke, B., Carleton, J., Skuza, M., Zagorskis, I., Furnas, M.J., 2012a. Water quality in the inshore Great Barrier Reef lagoon: Implications for long-term monitoring and management. Marine Pollution Bulletin 65, 249–260.

Schaffelke, B., J. Carleton, J. Doyle, M. Furnas, K. Gunn, M. Skuza, M. Wright, and I. Zagorskis. 2010. Reef Rescue Marine Monitoring Program. Final Report of AIMS Activities - Inshore water quality monitoring 2009/10. Australian Institute of Marine Science, Townsville. Available at: http://www.gbrmpa.gov.au/resources-and-publications/publications/annual-reef-rescue-marine-monitoring-science-report.

Schaffelke, B., Mellors, J. and Duke, N.C. 2005. Water quality in the Great Barrier Reef region: responses of mangrove, seagrass and macroalgal communities. Marine Pollution Bulletin 51, 279-296.

Schmidt, C., Heinz, P., Kucera, M., Uthicke, S., 2011. Bleaching in larger benthic foraminifera hosting endosymbiotic diatoms: effects of temperature induced stress. Limnology and Oceanography 56: 1287-1602

Schroeder, T., Dekker, A., Rathbone, C. 2009. Remote sensing for light attenuation mapping in the North Marine Region. CSIRO Wealth from Oceans Flagship Report to the Department of the Environment, Water, Heritage and the Arts CSIRO.

Shaw, J.F., Brodie. J. C. M., Müller, J.F., 2012. Phytotoxicity induced in isolated zooxanthellae by chemicals extracted from Great Barrier Reef flood waters. Marine Pollution Bulletin 65, 355-362.

Shaw, M., Furnas, M.J., Fabricius, K., Haynes, D., Carter, S., Eaglesham, G., Mueller, J.F., 2010. Monitoring pesticides in the Great Barrier Reef. Marine Pollution Bulletin 60, 113-122.

Shaw, M. Müller, J.F. 2005. Preliminary evaluation of the occurrence of herbicides and PAH's in the Wet Tropics region of the Great Barrier Reef, Australia, using passive samplers. Marine Pollution Bulletin 51, 876-881.

Shaw, C.M., Müller, J.F., Brodie, J., 2012. Phytotoxicity induced in isolated zooxanthellae by chemicals extracted from Great Barrier Reef flood waters. Marine Pollution Bulletin 65, 355–362.

Shaw, M., Negri, A., Fabricius, K., Mueller, J.F., 2009. Predicting water toxicity: Pairing passive sampling with bioassays on the Great Barrier Reef. Aquatic Toxicology 95, 108-116.

Sherman, R.E., Fahey, T.J. and Martinez, P. 2001. Hurricane impacts on a mangrove forest in the Dominican Republic, damage patterns and early recovery. Biotropica 33, 393-408.

Smith, R., Middlebrook, R., Turner, R., Huggins, R., Vardy, S., Warne, M. 2012. Large-scale pesticide monitoring across Great Barrier Reef catchments – Paddock to Reef Integrated Monitoring, Modelling and Reporting Program. Marine Pollution Bulletin 65, 117-127.

Sofonia, J., Anthony, K.R.N., 2008. High-sediment tolerance in the reef coral *Turbinaria mesenterina* from the inner Great Barrier Reef lagoon (Australia). Estuarine, Coastal and Shelf Science 78, 748–752

Stafford-Smith, M.G., Ormond, R.F.G., 1992. Sediment-rejection mechanisms of 42 species of Australian scleractinian corals. Australian Journal of Marine and Freshwater Research 43, 683-705.

Stearns, S.C., 1989. Trade-offs in life history evolution. Functional Ecology 3, 259-268.

Sweatman, H., Delean, S., Syms, C., 2011. Assessing loss of coral cover on Australia's Great Barrier Reef over two decades, with implications for longer-term trends. Coral Reefs 30, 521-531.

Sweatman, H., Syms, C., 2011. Assessing loss of coral cover on Australia's Great Barrier Reef: A response to Hughes, *et al.,* 2011. Coral Reefs 30, 661-664.

Te, F.T., 1997. Turbidity and its effect on corals: a model using the extinction coefficient, k. of photosynthetic active radiance., PAR.. Proceedings of the 8th International Coral Reef Symposium 2, 1899-1904.

Thompson, A., Costello, P., Davidson, J., Liddy, M., Schaffelke, B., Uthicke, S., 2012. Reef Rescue Marine Monitoring Program. Final Report of AIMS Activities 2012: Inshore coral reef monitoring. Report for Great Barrier Reef Marine Park Authority. Australian Institute of Marine Science, Townsville.

Thompson, A., Costello, P., Davidson, J., Logan, M., Schaffelke, B., Uthicke, S., Takahashi, M., 2011 Reef Rescue Marine Monitoring Program. Report of AIMS Activities – Inshore coral reef monitoring 2011. Report for Great Barrier Reef Marine Park Authority. Australian Institute of Marine Science, Townsville. 128 pp.

Thompson, A., Dolman, A., 2010. Coral bleaching: one disturbance too many for inshore reefs of the Great Barrier Reef. Coral Reefs 29, 637-648 .

Thompson, A., Schaffelke, B., De'ath, G., Cripps, E., Sweatman, H., 2010. Water Quality and Ecosystem Monitoring Programme-Reef Water Quality Protection Plan. Synthesis and spatial analysis of inshore monitoring data 2005-08. Report to the Great Barrier Reef Marine Park Authority. Australian Institute of Marine Science, Townsville.

Titlyanov, E.A., 1991. Light adaption and production characteristics of branches differing by age and illumination of the hermatypic coral *Pocillopora verrucosa*. Symbiosis 10, 249-260.

Todd, P.A., 2008. Morphological plasticity in Scleractinian corals. Biological Reviews 83, 315-337.

Tsatsaros, J.H., Brodie, J.E., Bohnet, I.C., Valentine, P., 2013. Water quality degradation of coastal waterways in the Wet Tropics, Australia. Water, Air and Soil Pollution 224, 1443.

Turner, R., Huggins, R., Wallace, R., Smith, R., Vardy, S., Warne, M. St. J., 2012. Sediment, Nutrient and Pesticide Loads: Great Barrier Reef Catchment Loads Monitoring 2009-2010, Department of Science, Information Technology, Innovation and the Arts, Brisbane. pp. 53.

Turner., R, Huggins., R, Wallace., R, Smith., R, Vardy., S, Warne., M St. J., in press. Loads of sediment, nutrients and pesticides discharged from high priority Queensland rivers in 2010-2011. Great Barrier Reef Catchment Loads Monitoring Program, Department of Science, Information Technology, Innovation and the Arts, Brisbane.

Turner, R.E., Rabalais, N.N., Nan, Z.Z. 1990. Phytoplankton biomass, production and growth limitations on the Huanghe (Yellow) River continental shelf. Continental Shelf Research 10, 545–571.

Udy, J.W., Dennison, W.C., 1997a. Physiological responses of seagrasses used to identify anthropogenic nutrient inputs. Marine and Freshwater Research 48, 605–614.

Udy, J.W., Dennison, W.C., 1997b. Growth and physiological responses of three seagrass species to elevated sediment nutrients. Journal of Experimental Marine Biology and Ecology 217, 253–277

Udy, J.W., Dennison, W.C., Lee Long, W.J., McKenzie, L.J., 1999. Responses of seagrass to nutrients in the Great Barrier Reef, Australia. Marine Ecology Progress Series 185, 257-271.

Uthicke, S., Altenrath, C., 2010. Water column nutrients control growth and C : N ratios of symbiont-bearing benthic foraminifera on the Great Barrier Reef, Australia. Limnology and Oceanography 55, 1681-1696

Uthicke, S., Llewellyn, L., Eder, F., 2009. A fluorescent lectin assay to quantify particulate marine polysaccharides on 96-well filtration plates. Limnology and Oceanography Methods 7, 449-458

Uthicke, S., Nobes, K., 2008. Benthic Foraminifera as ecological indicators for water quality on the Great Barrier Reef. Estuarine, Coastal and Shelf Science 78, 763-773.

Uthicke, S., Patel, F., Ditchburn, R., 2012a. Elevated land runoff after European settlement perturbs persistent foraminiferal assemblages on the Great Barrier Reef. Ecology 93, 111-121.

Uthicke, S., Thompson, A., Schaffelke, B., 2010. Effectiveness of benthic foraminiferal and coral assemblages as water quality indicators on inshore reefs of the Great Barrier Reef, Australia. Coral Reefs 29, 209-225.

Uthicke, S., Vogel, N., Doyle, J., Schmidt, C., Humphrey, C., 2012b. Interactive effects of climate change and eutrophication on the dinoflagellate-bearing benthic foraminifer *Marginopora vertebralis*. Coral Reefs 31, 401-414.

Valiela, I. and Cole, M.L. 2002. Comparative evidence that salt marshes and mangroves may protect seagrass meadows from land-derived nitrogen loads. Ecosystems 5, 92-102.

van Dam, J.W., Negri, A.P., Mueller, J.F., Altenburger, R., Uthicke, S., 2012a. Additive pressures of elevated sea surface temperatures and herbicides on symbiont-bearing foraminifera. PLoS ONE 7(3), e33900. Doi:10.1371/journal.pone.0033900.

van Dam, J.W., Negri, A.P., Mueller, J.F., Uthicke, S., 2012b. Symbiont-specific responses in foraminifera to the herbicide diuron. Marine Pollution Bulletin 65, 373-383.

van Katwijk, M.M., Bos, A.R., Kennis, P., de Vries, R., 2010. Vulnerability to eutrophication of a semi-annual life history: A lesson learnt from an extinct eelgrass (*Zostera marina*) population. Biological Conservation 143(1), 248-254.

van Katwijk, M.M.,Vergeer, L.H.T., Schmitz, G.H.W., Roelois, J.G.M., 1997. Ammonium toxicity in eelgrass *Zostera marina*. Marine Ecology Progress Series 157, 159–173

van Oppen, M., Baker, A., Coffroth, M.A., Willis, B., 2009. Bleaching resistance and the role of algal endosymbionts. In: van Oppen M.J.H, Lough J.M., eds. Coral Bleaching: Patterns, Processes, Causes and Consequences. Springer Heidelberg, pp 83-102.

van Woesik, R., Done, T.J., 1997. Coral communities and reef growth in the southern Great Barrier Reef. Coral Reefs 16:103-115.

van Woesik, R., Tomascik, T., Blake S., 1999. Coral assemblages and physico-chemical characteristics of the Whitsunday Islands: evidence of recent community changes. Marine and Freshwater Research 50, 427-440.

Verhoeven, J.T.A., Arheimer, B., Yin, C., Hefting, M.M. 2006. Regional and global concerns over wetlands and water quality. Trends in Ecology and Evolution 21, 96–103

Wake, J. 2005. Mangrove health in the Pioneer River Estuary. Centre for Environmental Management Central Queensland University Rockhampton Australia .

Wake, J. 2006. Interim-Resampling of Mangrove Health in the Pioneer River estuary2005/6. Centre for Environmental Management, Central Queensland University, Rockhampton Australia.

Wang, Y., Ridd, P.V., Heron, M.L., Stieglitz, T.C., Orpin, A.R. 2007. Flushing time of solutes and pollutants in the central Great Barrier Reef lagoon, Australia. Marine and Freshwater Research 58, 778-791.

Warne, M.St.J., Hawker, D.W., 1995. The number of components in a mixture determines whether synergistic and antagonistic or additive toxicity predominate: The funnel hypothesis. Ecotoxicology and Environmental Safety 31, 28-38.

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: J.E. Johnson, P.A. Marshall (eds). Climate Change and the Great Barrier Reef. Part II, Chapter 8. Great Barrier Reef Marine Park Authority and Australian Greenhouse Office, Australia. pp.193-235.

Waycott, M., Longstaff, B.J., Mellors, J., 2005. Seagrass population dynamics and water quality in the Great Barrier Reef region: A review and future research directions. Marine Pollution Bulletin 51, 343-350.

Waycott, M., McKenzie, L., Mellors, J., Ellison, J., Sheaves, M., Collier, C., Schwarz, A., Webb, A., Johnson, J. Payri, C., 2011. Vulnerability of mangroves, seagrasses and intertidal flats in the tropical Pacific to climate change. In: J. Bell, J. Johnson (eds). Vulnerability of fisheries and aquaculture in the Pacific to climate change. Chapter 6. Secretariat of the Pacific Community, Noumea, New Caledonia. pp97-168.

Waycott, M., McKenzie, L.J., 2010. Final report project 1.1.3 and 3.7.1 to the Marine and Tropical Sciences Research Facility: Indicators and thresholds of concern for seagrass ecosystem health for the Great Barrier Reef, Reef and Rainforest Research Centre Limited, Cairns, Australia

Weber, M., de Beer, D., Lott, C., Polerecky, L., Kohls, K., Abed, R., Ferdelman, T., Fabricius, K., 2012. A series of microbial processes kills corals exposed to organic-rich sediments. Proceedings of the National Academy of Sciences of the United States of America 109, E1558-E1567.

Weber, M., Lott, C., Fabricius, K., 2006. Different levels of sedimentation stress in a scleractinian coral exposed to terrestrial and marine sediments with contrasting physical, geochemical and organic properties. Journal of Experimental Marine Biology and Ecology 336, 18-32.

Webster, I.T., and Ford, P.W. 2010. Delivery, deposition and redistribution of fine sediments within macrotidal Fitzroy Estuary/Keppel Bay: southern Great Barrier Reef, Australia. Continental Shelf Research 30, 793 – 805.

Webster, N.S., Smith, L.D., Heyward, A.J., Watts, J.E.M., Webb, R.I., Blackall, L.L., Negri, A.P., 2004. Metamorphosis of a scleractinian coral in response to microbial biofilms. Applied and Environmental Microbiology 70, 1213-1221.

West, J.M., Salm, R.V., 2003. Resistance and resilience to coral bleaching: implications for coral reef conservation and management. Conservation Biology 17, 956-967.

Wetzel, R.G. 2001. Fundamental processes within natural and constructed wetland ecosystems: short-term versus long-term objectives. Water, Science and Technology 44, 1-8.

Wiedenmann, J., D'Angelo, C., Smith, E.G., Hunt, A.N., Legiret, F.E., Postle, A.D., Achterberg, E.P., 2012. Nutrient enrichment can increase the susceptibility of reef corals to bleaching. Nature Climate Change 3, 160-164.

Wolanski, E., van Senden, D., 1983. Mixing of Burdekin river flood waters in the Great Barrier Reef. Australian Journal of Marine and Freshwater Research 34, 49–63.

Wolanski, E. 1992. Mangrove hydrodynamics. In: Robertson, A.I. and Alongi, D.M. (Eds.), Tropical mangrove ecosystems. American Geophysical Union, Washington DC. pp. 43-62.

Wolanski, E. Fabricius, K. Cooper, T. Humphrey, C. 2008. Wet season fine sediment dynamics on the inner shelf of the Great Barrier Reef. Estuarine, Coastal and Shelf Science 77, 755-762.

Wolanski, E., 2007. Estuarine Ecohydrology. Elsevier, 168 p.

Wolanski, E., Fabricius, K.E., Spagnol, S., Brinkman, R., 2005. Fine sediment budget on an inner-shelf coral-fringed island, Great Barrier Reef of Australia. Estuarine, Coastal and Shelf Science 65, 153-158.

Wolanski, E., Jones, M. 1981. Physical properties of Great Barrier Reef lagoon waters near Townsville. I. Effects of Burdekin River Floods. Australian Journal of Marine and Freshwater Research 32, 305-319.

Woodroffe, C.C. and Grime, D. 1999. Storm impact and evolution of a mangrove fringed chenier plain, Shoal Bay, Darwin, Australia. Marine Geology 159, 303-321.

Woodroffe, C.D. 1992. Mangrove sediments and geomorphology. In: Robertson, A.I. and Alongi, D.M. (Eds.), Tropical mangrove ecosystems. American Geophysical Union, Washington DC. pp. 7-42.

Wooldridge, S., Brodie, J., Furnas, M., 2006. Exposure of inner-shelf reefs to nutrient enriched runoff entering the Great Barrier Reef Lagoon: Post-European changes and the design of water quality targets. Marine Pollution Bulletin 52, 1467-1479.

Wooldridge, S.A., 2009. Water quality and coral bleaching thresholds: Formalising the linkage for the inshore reefs of the Great Barrier Reef, Australia. Marine Pollution Bulletin 58, 745-751.

Wooldridge, S.A., Done, T.J., 2009. Improved water quality can ameliorate effects of climate change on corals. Ecological Applications 19, 1492-1499.