2017 Scientific Consensus Statement

CHAPTER ONE

The condition of coastal and marine ecosystems of the Great Barrier Reef and their responses to water quality and disturbances.

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

Contents

Acronyms, units and definitions

Acronyms

DIN = dissolved inorganic nitrogen

ENSO = El Niño Southern Oscillation

GBR = Great Barrier Reef

IPCC = Intergovernmental Panel on Climate Change

NESP = National Environmental Science Programme nesptropical.edu.au/

NRM = natural resource management

PSII = Photosystem II (water-plastoquinone oxidoreductase) is the first protein complex in the light-dependent reactions of oxygenic photosynthesis

RCP = Representative Concentration Pathways

SST = sea surface temperature

TC = tropical cyclone

TSS = total suspended sediment[1]

Units

°C = degrees Celsius

km^2 = square kilometres

mm = millimetres

mm/yr = millimetres per year

$mol/m^2/d$ = moles of light per square metre per day

µg/L = micrograms per litre

µL/L = microlitres per litre

µm = micrometres (microns)

t = tonnes

Definitions

Basin: There are 35 basins that drain into the Great Barrier Reef. A basin can be made up of a single or multiple rivers (e.g. North and South Johnstone is one basin). Basins are primarily used here when discussing the relative delivery of a pollutant to the marine system.

Catchment: The natural drainage area upstream of a point that is generally on the coast. It generally refers to the 'hydrological' boundary and is the term used when referring to modelling in this document. There may be multiple catchments in a basin.

Pollutants: Pollution means the introduction by humans, directly or indirectly, of substances or energy into the environment resulting in such deleterious effects as harm to living resources, hazards to human health, hindrance to aquatic activities including fishing, impairment of quality for use of water and reduction of amenities (GESAMP, 2001). This document refers to suspended (fine) sediments, nutrients (nitrogen, phosphorus) and pesticides as 'pollutants'. Within this chapter we

[1] TSS is also often referred to as total suspended solids.

explicitly mean enhanced concentrations of or exposures to these pollutants, which are derived from (directly or indirectly) human activities in the Great Barrier Reef ecosystem or adjoining systems (e.g. river catchments). Suspended sediments and nutrients naturally occur in the environment; all living things in ecosystems of the Great Barrier Reef require nutrients, and many have evolved to live in or on sediment.

Region: There are six natural resource management (NRM) regions covering the Great Barrier Reef catchments. Each region groups and represents catchments with similar climate and bioregional setting, with boundaries extending into the adjacent marine area. The regions are Cape York, Wet Tropics, Burdekin, Mackay Whitsunday, Fitzroy and Burnett Mary.

Wetlands:[2] areas of permanent or periodic/intermittent inundation, with water that is static or flowing fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed 6m. To be a wetland the area must have one or more of the following attributes:

- At least periodically, the land supports plants or animals that are adapted to and dependent on living in wet conditions for at least part of their life cycle.
- The substratum is predominantly undrained soils that are saturated, flooded or ponded long enough to develop anaerobic conditions in the upper layers.
- The substratum is not soil and is saturated with water, or covered by water at some time.

Wetland systems include:[3]

- *Riverine wetlands* are all wetlands and deepwater habitats within a channel. The channels are naturally or artificially created, periodically or continuously contain moving water or connect two bodies of standing water.
- *Palustrine wetlands* are primarily vegetated non-channel environments of less than 8ha. They include billabongs, swamps, bogs, springs, soaks etc. and have more than 30% emergent vegetation.
- *Estuarine wetlands* are those with oceanic water sometimes diluted with freshwater run-off from the land.
- *Lacustrine wetlands* are large, open, water-dominated systems (e.g. lakes) larger than 8ha. This definition also applies to modified systems (e.g. dams), which are similar to lacustrine systems (e.g. deep, standing or slow-moving waters).

Acknowledgements

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[2] Queensland Wetlands Program wetland definition: https://wetlandinfo.ehp.qld.gov.au/wetlands/what-are-wetlands/definitions-classification/wetland-definition.html

[3] Wetland systems, WetlandInfo, Department of Environment and Heritage Protection, Queensland, viewed 7 April 2017, https://wetlandinfo.ehp.qld.gov.au/wetlands/what-are-wetlands/definitions-classification/system-definitions.html

Dieter Tracey (TropWATER, James Cook University) is gratefully acknowledged for the preparation of the overview map and the cyclone and river discharge figures. Significant contributions to content in this chapter were also provided by Dr Fernanda Adame, Australian Rivers Institute, Griffith University and Amelia Selles, Wetlands Team, Department of Environment and Heritage Protection. Thank you to Jane Mellors (TropWATER James Cook University) for final editing.

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Executive summary

The Great Barrier Reef marine ecosystems and their associated catchments are part of a dynamic, interconnected system. This chapter provides an up-to-date review of the state of knowledge relating to the conditions and trends of key Great Barrier Reef coastal and marine ecosystems, including current knowledge on key drivers of change and activities leading to pressures and impacts on these ecosystems. Drivers include the impacts of land run-off, coastal development activities and other disturbances such as extreme weather events that influence Great Barrier Reef water quality and the health of marine and coastal ecosystems.

The condition of inshore coral reefs and seagrass meadows marginally improved between 2014 and 2015 but is still rated moderate to poor, depending on the region. The improvement coincided with a period of low tropical cyclonic disturbance and no significant river flood events affecting the Great Barrier Reef between 2014 and 2016. However, in 2016 record warm sea surface temperatures caused a mass coral bleaching event and significant coral mortality, especially on reefs north of Port Douglas. The condition of mid-shelf and outer shelf coral reefs in the Cairns/Cooktown Management Area also continues to be affected by a population outbreak of crown-of-thorns starfish, arguably the most important indirect effect of excess nutrients on Great Barrier Reef coral reefs.

Research over recent years has improved the understanding of interactive and cumulative pressures as drivers of coastal and marine ecosystem condition and resilience. The global pressures of ocean warming and acidification and local disturbances by tropical cyclones interact with local, anthropogenic pressures such as land run-off. However, predicting conditions of coastal and marine ecosystems is difficult due to the challenge of untangling and quantifying individual pressure–response relationships, non-linear responses and the complexity of interactions and ecological feedback loops in a changing environment. The main findings of the chapter are summarised here around four key questions:

What are the conditions and trends of water quality in the coastal and marine areas of the Great Barrier Reef region?

- Water quality in the inshore waters from the Wet Tropics to the Fitzroy regions remains in moderate to poor condition and is strongly associated with increased land-based run-off.
- The increase in land-based run-off since European settlement has now been confirmed in coral cores for sediment, nitrogen and phosphorus and is additionally associated with a climatic shift to more variable and more frequent extreme river flood events.
- Turbidity and photic depth in the Great Barrier Reef lagoon are significantly affected by rainfall and river flow, as well as by wave and tidal effects.
- Monitoring revealed a wide range of photosystem II herbicides and other pesticides (e.g. terbutryn, imidacloprid, imazapic and metolachlor) in Great Barrier Reef coastal and marine ecosystems.
- The half-lives of widely used photosystem II herbicides are more than 100 days, indicating high persistence and explaining their presence in the Great Barrier Reef year-round.
- Contaminants such as antifouling paint components, coal dust and particles, marine debris (including microplastics), metals and metalloids, petroleum hydrocarbons, pharmaceuticals and personal care products are present in Great Barrier Reef coastal and/or marine ecosystems.

What are the conditions and trends of key coastal and marine ecosystems of the Great Barrier Reef region?

Coastal freshwater wetlands and estuarine ecosystems

- In the catchments of the Great Barrier Reef, there are over 15,000 km^2 of mapped wetlands (artificial/highly modified, lacustrine, palustrine, riverine and estuarine).
- The extent of natural and near-natural wetlands has largely been maintained since the previous Scientific Consensus Statement at around 85% of pre-European extent, based on the most recent mapping assessment in 2013.
- The extent of mapped freshwater swamps (palustrine wetlands) remains at 76% of pre-European extent, with an overall loss of 59 ha between 2009 and 2013.
- However, historic loss of floodplain wetlands exceeds 80% of pre-European extent in the lowland areas of some catchments, such as the Herbert.
- Over 90% of pre-European extent of mangroves and saltmarshes remain in most catchments, except for the Barron, Mulgrave, Herbert, Bohle, Ross, Lower Burdekin, Pioneer, Calliope, Boyne, Fitzroy, Shoalwater, Burnett and Kolan catchments, with an overall loss of 293 ha between 2009 and 2013.
- Most of the recent net loss of wetlands is a result of the loss of saltmarsh wetlands in the Calliope catchment associated with industrial and port development at Gladstone.
- Remnant wetlands in the Great Barrier Reef catchment maintain high biodiversity and aesthetic and cultural values and provide important connections between freshwater and marine ecosystems of the Great Barrier Reef region and ecosystem services such as water quality improvement and carbon storage.
- Wetlands may have a filtering capacity to remove land-based pollutants and contribute to water quality improvement; however, these pollutants can also have significant negative impacts on wetlands.

Seagrass ecosystems

- Inshore seagrass meadows remain in poor condition, despite improvements in some seagrass condition indicators in some regions. There were overall improvements in abundance (above-ground per cent cover and biomass); however, reproductive effort declined, indicating a low capacity to recover from disturbances with the available seed resources.

Coral reef ecosystems

- The condition of inshore reefs (Wet Tropics to Fitzroy region) has marginally improved between 2014 and 2015 but is still rated moderate to poor, depending on the region. The improvement coincided with a period of low disturbance, after a decline in condition during the 2007-2013 period of very wet years and tropical cyclone disturbances.
- Hard coral cover on mid-shelf and offshore reefs south of around Cairns increased between 2012 and 2015, showing very fast recovery during a period of low disturbance. Coral cover declined in the Cairns/Cooktown Management Area, associated with the impact of tropical cyclones and crown-of-thorns starfish outbreaks.
- A mass thermal coral bleaching event affected the Great Barrier Reef in 2016 and resulted in significant coral mortality, especially north of Port Douglas. The inshore reefs monitored as part of the Marine Monitoring Program (Wet Tropics to Fitzroy region) appear to have escaped major coral mortality.

What are the key drivers of the observed changes?

- Several tropical cyclones affected the Great Barrier Reef between 2013 and 2015, with two severe cyclones in the area between Lockhart River and Cooktown in 2014 and 2015, respectively, and one severe cyclone crossing the Mackay/Capricorn Management Area. No cyclones were recorded in the Great Barrier Reef region in 2016.

- Sea surface temperatures of the Great Barrier Reef have significantly warmed since the late 19th century by 0.80°C and will continue to warm.
- Record warm sea surface temperatures were observed on the entire Great Barrier Reef in March, April and May 2016. Temperatures in the northern half of the Great Barrier Reef remained extremely high into late summer and autumn, while temperatures further south were slightly moderated in February 2016 by two tropical lows bringing cloud cover, rain and wind.
- Paleoclimate records highlight the unusual nature of recent warmth and also provide evidence of more frequent extreme rainfall and river flows affecting the Great Barrier Reef.
- Climate models predict continued warming; increasing intensity of extreme rainfall events, resulting in freshwater floods; fewer but more intense tropical cyclones; and more frequent and extreme La Niña and El Niño events.
- The 2013 water year (October 2012 to September 2013) was the last in a sequence of very wet years (2007-2013), with total river discharge into the Great Barrier Reef of more than twice the long-term average. Total river discharge in the 2014-2016 water years was around or below average and no significant river flood events affected the Great Barrier Reef lagoon.
- The condition of many coastal freshwater wetlands continues to be affected by a range of chronic and acute pressures such as excess nutrient, sediment and pesticide loads, loss of connectivity, changes in hydrology, and invasive species. These modifications have led to secondary impacts such as algal blooms and hypoxia.
- Mangroves in the Great Barrier Reef region are at relatively low risk of submersion related to sea level rise; however, estuarine wetland composition will be affected as mangroves lose their fringing forest and potentially migrate landwards into saltmarsh and/or adjacent freshwater wetlands.
- Mangroves and saltmarshes continue to be affected by excess nutrient, sediment and pesticide loads, especially during periods of high rainfall. Degradation due to tropical storms has been observed in the Hinchinbrook Channel.
- Light is one of the most critical variables affecting seagrass growth and survival. Pulses of pollutants from river run-off containing suspended sediments, nutrients (which can trigger phytoplankton blooms) and coloured dissolved organic matter reduce the levels of light reaching seagrass. The impact of reduced light on seagrass is exacerbated by higher seawater temperatures.
- In large-scale flood events, seagrass health can be directly affected by sediment deposition close to river mouths; however, of greater concern is the legacy of finer sediments in floodwaters, which continually resuspend (for at least six to eight months) and hinder recovery during the growth season.
- Despite decreases in nutrients (nitrogen and phosphorus) within seagrass habitats since 2010-2011, the ratio of carbon to nitrogen indicates that nitrogen loads remain elevated compared to biological demand. At subtidal sites there have been increasing epiphyte loads since 2014, potentially indicating elevated nutrients. These, together with high frequency of exposure to green, phytoplankton-rich water, are indirect impacts of nutrient loading to inshore seagrass meadows.
- Many inshore seagrass meadows of the Great Barrier Reef are exposed to herbicide concentrations year-round, leaving the plants vulnerable to other simultaneous stressors such as reduced light, but the contribution of herbicides to annual changes in seagrass abundance and to widespread seagrass losses between 2009 and 2011 is unknown.
- Turbidity resulting from high concentrations of suspended sediments, and sedimentation, are two of the most widely recognised threats to coral reefs.
- The fourth crown-of-thorns starfish population outbreak commenced around 2010-2011 and has now progressed south, to reefs at the latitude of about Ingham. Currently, the most widely

accepted hypothesis is that primary outbreaks are promoted by high nutrient availability, such as are observed after significant flood events, which increase larval food supply and survivorship. Recent research has supported this nutrient-limitation hypothesis, but some uncertainty about the exact chlorophyll-based threshold exists. There is some evidence for other factors contributing to outbreaks, such as removal of predators of adult crown-of-thorns starfish, retention of crown-of-thorns starfish larvae in the outbreak initiation region in the northern Great Barrier Reef and higher temperatures increasing larval survivorship.

Are the ecosystems showing signs of resilience to environmental pressures and changes?

- The resilience of Great Barrier Reef coastal and marine ecosystems, that is resistance to pressures and ability to recover during periods of low disturbance, is affected by the cumulative impacts of global pressures, in concert with local disturbances (e.g. tropical cyclones) and local, anthropogenic pressures such as land run-off.
- The key management approach in the Reef 2050 Long-Term Sustainability Plan is the mitigation of local stressors to support ecosystem resilience and 'buy time' for the mitigation of global pressures and for organisms to adapt to a changing environment. The understanding of acclimatisation and adaptation rates of Great Barrier Reef organisms is currently poor.
- Limited information exists on the recovery and resilience of coastal wetlands and estuarine ecosystems, including mangroves, because their condition is not systematically monitored. Some anecdotal information exists on recovery for individual wetlands (e.g. Mungalla in the Herbert Basin).
- In overseas studies wetlands have been shown to be resilient to increases in nutrient loads, as they can process them; however, their resilience is limited and once the nutrient levels reach an upper limit the function of the wetland may be irreversibly changed.
- At present, Great Barrier Reef seagrass meadows continue to recover from losses experienced during the adverse weather of 2009-2011. Recovery rates vary spatially and temporally due to varying stressors, reproductive output and availability of seed banks. Seagrass recovery was hindered in some regions of the Great Barrier Reef with the concomitant loss of ecosystem engineering functions (e.g. seagrasses stabilising the seabed thereby reducing sediment erosion or resuspension), undermining habitat resilience.
- Ecological aspects of recovery in seagrass (e.g. changes in species composition) have been documented through monitoring, but recovery processes (e.g. triggers for seed germination, seed viability, seed bank thresholds, sediment conditions, species interactions) remain critical information gaps that preclude accurate prediction of recovery rates for Great Barrier Reef seagrass communities.
- Coral reefs are sensitive to disturbances such as cyclones and crown-of-thorns starfish outbreaks and to environmental extremes such as high temperatures and major floods. The mid-shelf and outer shelf reefs of the Great Barrier Reef have shown capacity to rapidly recover from recent disturbances, mostly through increase in cover of fast-growing coral species. The recent period of low rainfall and run-off demonstrates the inherent ability of inshore reefs communities to recover from acute disturbances during periods of reduced catchment loads.
- An example of the value of mitigation of local pressures are no-take zones on Great Barrier Reef reefs; these have been associated with greater resistance to disturbance and shorter recovery times of coral cover and reef fish communities after disturbance.
- Recent research provides more information on the response of resilience-related attributes: the exposure to reduced water quality influences the energetic and physiological status of corals, which may decrease their resistance to high temperature events; coral fertilisation and larval settlement, as important aspects of recovery, are vulnerable to suspended sediment and sedimentation, while pelagic embryo and larval stages appear to be more tolerant.

Key knowledge gaps and research recommendations

There is still appreciable uncertainty in our knowledge of the responses of coastal and marine ecosystems to the cumulative impacts of multiple pressures. To improve assessments of coastal and marine water quality and to attribute responses of ecosystems to critical drivers of ecological changes it is important to:

- increase water quality observations to improve the validation and calibration of the eReefs biogeochemical model and to refine existing water quality guideline values
- determine the transformation and bioavailability of terrestrially sourced particulate and organic nitrogen in the marine environment
- improve understanding of the transport, fate and impacts of very fine sediments and of organics flocs
- quantify tolerance thresholds and tipping points in key freshwater floodplain, seagrass and coral reef species and communities in response to single and multiple pressures
- improve understanding of recovery processes of seagrass meadows and coral reefs
- develop spatial ecosystem models to predict future ecosystem condition, resilience and recovery, with an immediate focus on coral reefs and seagrass meadows
- define desired states of seagrass meadows and coral reefs so that ecologically relevant targets can be set for the management of water quality-related pressures
- improve understanding of the scope and rates of acclimatisation and adaptation of coral reef taxa and seagrasses to environmental change, in particular a changing climate
- improve understanding of the mechanisms and processes by which nutrient run-off promotes crown-of-thorns starfish outbreaks to better focus management investment that will mitigate crown-of-thorns starfish impacts
- prioritise freshwater and estuarine barriers for mitigation or removal to improve water quality and fish passage in coastal ecosystems.

1. Introduction

The iconic Great Barrier Reef is the world's largest tropical coral reef system, comprising approximately 3000 reefs and extending over 2,000 km along the Queensland coast. The Great Barrier Reef was proclaimed a Marine Park in 1975 and listed on the World Heritage Register in 1981.

The Great Barrier Reef marine ecosystems are interconnected with the Coral Sea on the ocean side, and the freshwater and estuarine catchments and ecosystems on the landside (Waterhouse et al., 2016) (Figure 1).

The key conclusions in the previous 2013 Scientific Consensus Statement were that the marine water quality of the Great Barrier Reef continued to be negatively affected by the discharge of excess nutrients, fine sediments and pesticides from the adjacent catchments. During the period from 2008 to 2013 poor marine water quality was concluded to be a major cause of the poor state of many Great Barrier Reef ecosystems (coral reefs, seagrass meadows, coastal wetlands and estuaries), exacerbated by extreme weather such as heavy rainfall, floods and tropical cyclones. The conclusions of the 2013 Scientific Consensus Statement were supported by the Great Barrier Reef Marine Monitoring Program and by laboratory and field studies.

Since 2013, five major comprehensive assessments of the condition of Great Barrier Reef coastal and marine ecosystems were completed; the assessments also examined the risk of current and potential pressures to the ecosystems' long-term resilience. The assessments were the previous Scientific Consensus Statement (Brodie et al., 2013), the Great Barrier Reef Outlook Report (GBRMPA, 2014a), the Great Barrier Reef Region Strategic Assessment, jointly conducted by the Queensland and Australian governments (Queensland Department of State Development Infrastructure and Planning, 2013; GBRMPA, 2014b) and the Final Report of the Great Barrier Reef Water Science Taskforce (GBRWST, 2016). In addition, all six natural resource management (NRM) regions bordering the Great Barrier Reef have completed an update of their Water Quality Improvement Plans. These plans include a detailed section with a description of the regional coastal and marine ecosystems in each of these regions and an assessment of their condition (Folkers et al., 2014; Burnett Mary NRM Group, 2015; Fitzroy Basin Association, 2015; Terrain NRM, 2015; Cape York NRM and South Cape York Catchments, 2016; NQ Dry Tropics, 2016). The consensus of these comprehensive assessments was that the condition of coastal aquatic and marine ecosystems of the Great Barrier Reef is affected by the cumulative impacts of local and global pressures, specifically:

- the ongoing pressure from coastal land uses (e.g. clearing and modifying natural coastal habitats for agricultural, urban, industrial and island development, as well as illegal fishing activities)
- the ongoing pressure from land-based run-off of nutrients, sediments, pesticides and marine debris
- the long-term and increasing risk of climate change pressures, resulting in warming sea temperatures, ocean acidification, sea level rise and more extreme weather events.

In this chapter, we review and synthesise published information to provide an update of the conditions and trends of key Great Barrier Reef coastal aquatic and marine ecosystems, that is, coastal freshwater and estuarine wetlands, seagrass meadows and coral reefs (Figure 1).[4] We assimilate and present current knowledge on key drivers of change and activities leading to pressures and impacts on these ecosystems. Drivers include the impacts of land run-off, coastal

[4] Note: This chapter provides an update of two chapters in the 2013 Scientific Consensus Statement: Chapter 1 'Marine and coastal ecosystem impacts' and Chapter 2 'Resilience of Great Barrier Reef ecosystems and drivers of change'.

development activities and other disturbances such as extreme weather events that influence Great Barrier Reef water quality and the health of marine and costal ecosystems. The chapter focuses on the re-evaluation of the following overarching conclusions of the 2013 Scientific Consensus Statement against the new information:

- The decline of marine water quality associated with terrestrial run-off from the adjacent catchments is a major cause of the current poor state of many of the key marine ecosystems of the Great Barrier Reef.

Recent extreme weather—heavy rainfall, floods and tropical cyclones—have severely impacted marine water quality and Great Barrier Reef ecosystems. Ongoing climate change is predicted to increase the intensity of extreme weather events.

In addition, this chapter provides more detail on the state of knowledge of emerging pollutants that were not covered in the 2013 Scientific Consensus Statement and on coastal freshwater wetlands.

1.1 Synthesis process

This chapter has been co-written by a group of scientists, all of whom are active researchers with scientific expertise and a track record of published research on the condition, responses and pressures affecting change in coastal and marine ecosystems in the Great Barrier Reef region.

The contributors to this chapter conducted a comprehensive review of published information since 2013, the year of the last Scientific Consensus Statement, accessing the latest scientific information available in peer-reviewed scientific journals and technical reports. The geographic scope is the Great Barrier Reef region with a focus on coastal and inshore marine areas, plus Torres Strait recognising the connectedness of this region with the Great Barrier Reef Marine Park but acknowledging that very limited information is available for some areas.

The review focused on peer-reviewed research, published or available in online resources at the time of writing. To achieve the highest level of transparency, information that was unpublished or not readily accessible to a wider audience was not included in the update of current scientific information in this chapter.

1.2 What questions this chapter will answer

By the end of this chapter, readers should be able to answer the following questions:

- What are the conditions and trends of water quality and other key environmental indicators in the coastal and marine areas of the Great Barrier Reef region?
- What are the conditions and trends of key coastal and marine ecosystems of the Great Barrier Reef region?
- What are the key drivers of observed changes, considering human activities in the catchment and the coastal zone as well as climate change and extreme weather events?
- Are the ecosystems showing signs of resilience to environmental pressures and changes?

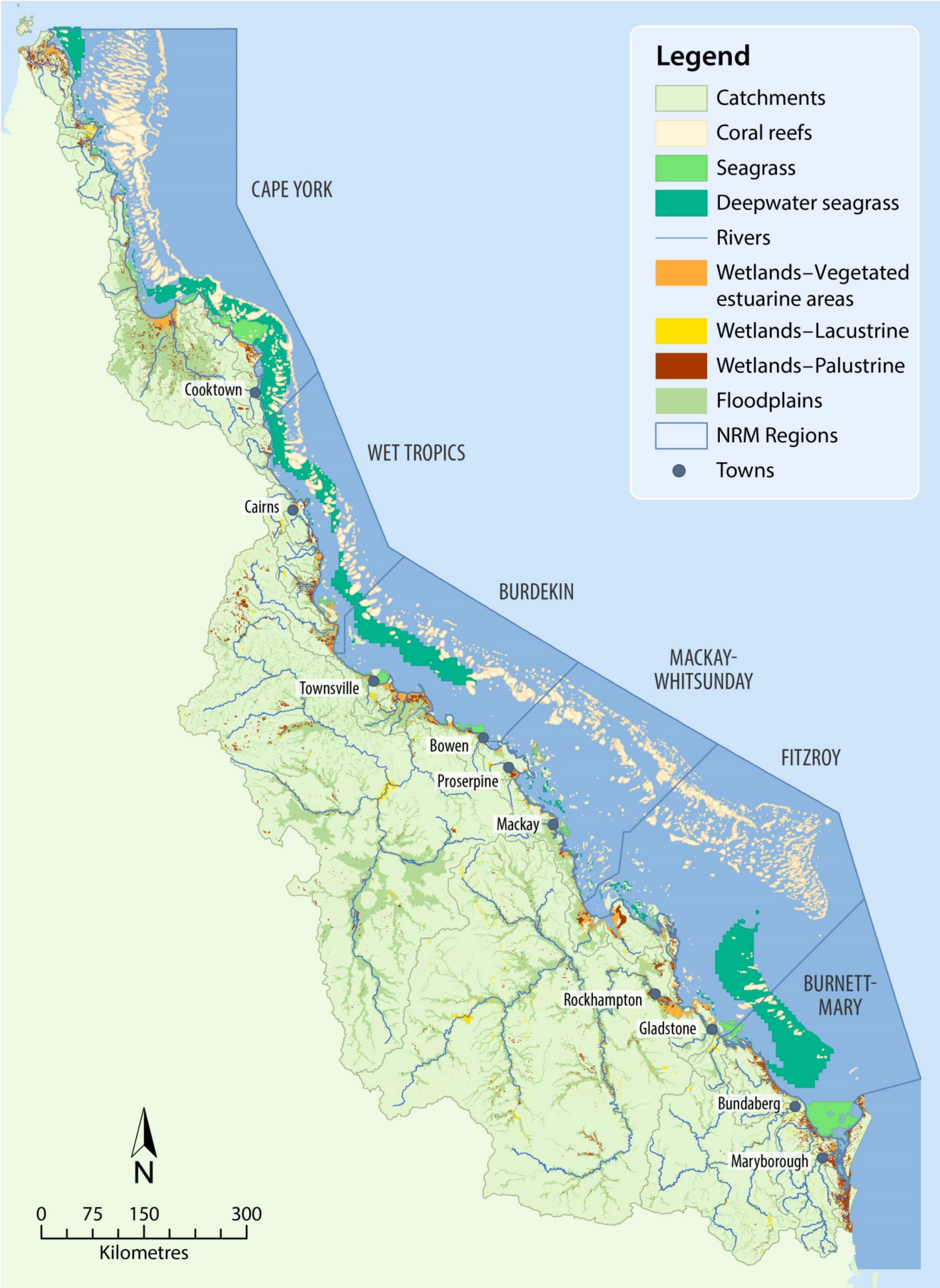

Figure 1: Geographic distribution of the ecosystems reviewed in this chapter and relevant administrative boundaries.

1.3 Disturbances and extreme weather/climate events affecting the coastal and marine areas of the Great Barrier Reef region

1.3.1 Flood events 2013-2016

The 2013 water year (October 2012 to September 2013) was the last in a sequence of very wet years (2007-2013), with total river discharge into the Great Barrier Reef of more than twice the long-term average. Total river discharge in the 2014 to 2016 water years was close to or below average and no significant river flood events affected the Great Barrier Reef lagoon (Figure 2).

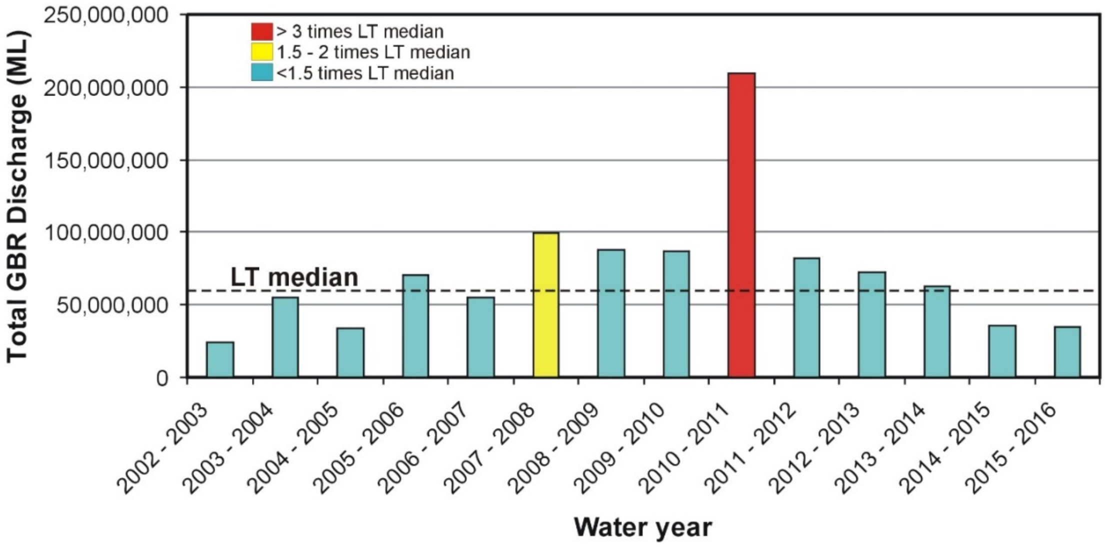

Figure 2: Long-term total discharge from the 35 main rivers draining into the Great Barrier Reef lagoon in million litres (ML) per water year (1 October to 30 September). Data from the Queensland Department of Resources and Mines (watermonitoring.dnrm.qld.gov.au/host.htm). (Figure from Waterhouse et al., in review)

1.3.2 Tropical cyclones 2013-2016

Several tropical cyclones (TC) affected the Great Barrier Reef from 2013 to 2016 (Figure 3). In this period two severe cyclones affected the northern Great Barrier Reef between Lockhart River and Cooktown: TC Ita (in 2014, maximum category 5) and TC Nathan (in 2015, maximum category 4). A third severe cyclone, TC Marcia (in 2015, maximum category 5), affected the Mackay/Capricorn Management Area. No cyclones were recorded in the region in 2016. However, in 2016 the tropical lows associated with ex-tropical cyclones Tatiana and Winston crossed into the Great Barrier Reef lagoon (Figure 4) and resulted in significant cooling of surface waters (see below).

Tropical cyclones can cause substantial structural damage to coastal and marine ecosystems, depending on their intensity, size and duration (Puotinen et al., 2016). For example, the most extreme force of the very strong (category 5) and large TC Yasi in 2011 caused severe and widespread damage to coral reefs (Beeden et al., 2015) and seagrass meadows (Pollard and Greenway, 2013; Rasheed et al., 2014). On inshore reefs, the observed structural and geomorphic changes, including sediment deposition, were highly site-specific (Perry et al., 2014). An analysis of three recent category 5 cyclones showed significant declines in coral cover and in the species richness and abundance of associated fish communities on reefs south of about Cairns by TC Hamish in 2009 and TC Yasi in 2011 and less severe declines on reefs north of Cairns by TC Ita in 2014 (Cheal et al., 2017).

Tropical cyclones are more likely to lead to extreme rainfall during La Niña years. Tropical cyclones contribute 20–30% of average summer rainfall and about 35% of annual rainfall maxima along the Queensland coast (Khouakhi et al., 2017).

2013 **2014** **2015** **2016**

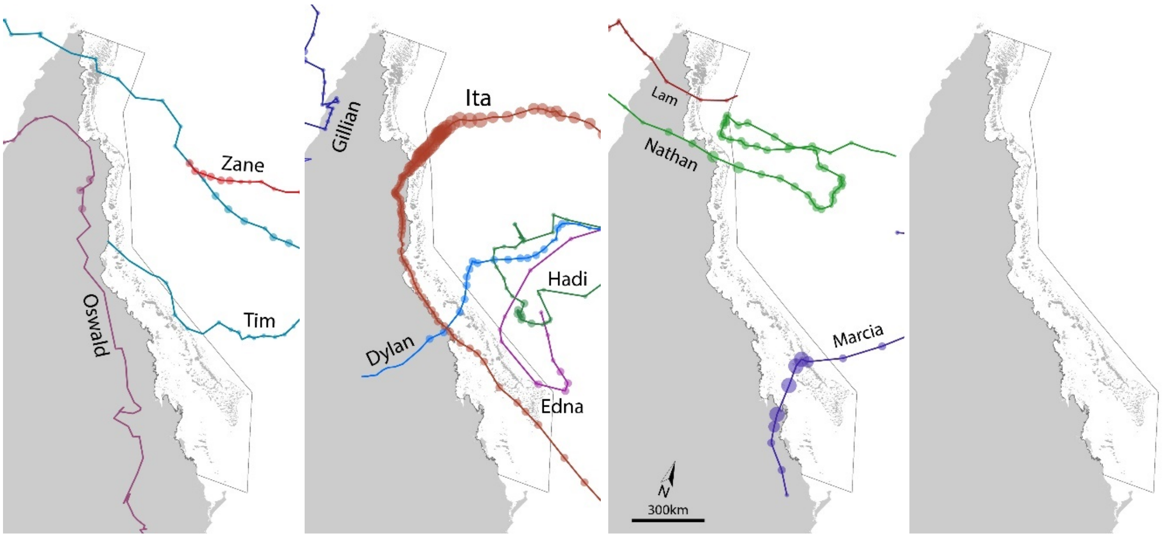

Figure 3: Tracks of tropical cyclones affecting the Great Barrier Reef between 2013 and 2016. Nine of these cyclones were category 3 or above and have affected the health of the Great Barrier Reef. All of the category 5 cyclones that affected the Great Barrier Reef since 1970 have occurred in the last decade (including TCs Larry, Hamish, Yasi, Ita and Marcia). Many of these cyclones have caused widespread flooding from intensive rainfall events in many parts of the Great Barrier Reef catchment. Note only tracks of cyclones that affected the Great Barrier Reef region are named.

Ex-tropical cyclone Tatiana **Ex-tropical cyclone Winston**

Figure 4: Tracks of two ex-tropical cyclones affecting the Great Barrier Reef as tropical lows in February 2016. (Maps from www.australiasevereweather.com/tropical_cyclones/oper_2015_2016_australian_region_tropical_cyclones.htm)

1.4 High temperature event in 2016

Record warm sea surface temperatures were observed throughout the Great Barrier Reef lagoon in March, April and May 2016 (e.g. Figure 5). Temperatures in the central and southern Great Barrier Reef were slightly moderated in February 2016 by two tropical lows (Figure 4) that brought cloud cover, rain and wind, while temperatures in the northern Great Barrier Reef remained extremely high into late summer and autumn. The high temperatures on the Great Barrier Reef were associated with the major 2015-2016 El Niño event that led to high temperatures in tropical oceans worldwide and resulted in a global, multi-year coral bleaching event that appears to be ongoing in some regions of the world[5] (Eakin et al., 2016). The Great Barrier Reef is also experiencing widespread bleaching again in the summer of 2017.[6]

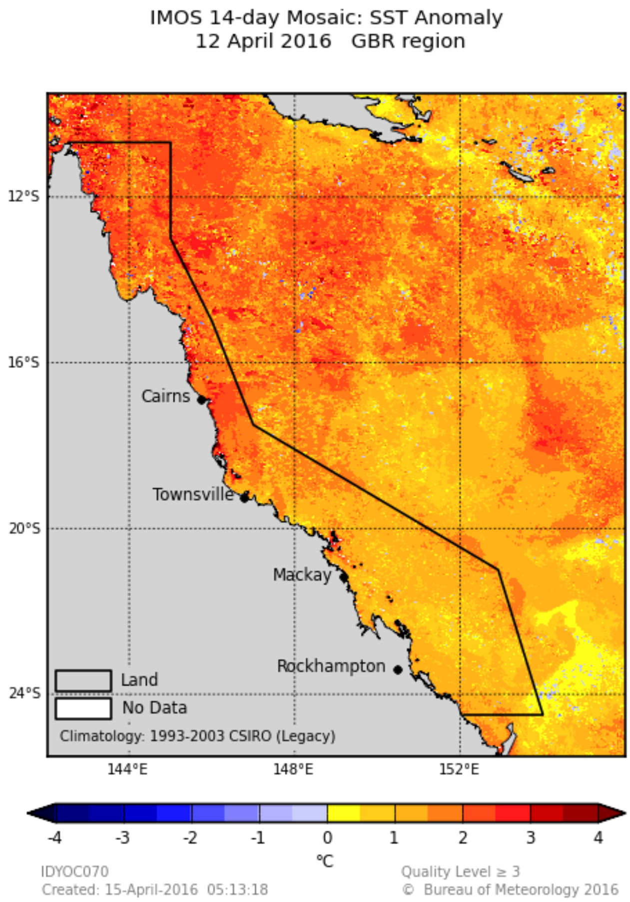

Figure 5: Sea surface temperature anomaly in April 2016, showing strongly elevated temperatures in the Great Barrier Reef Marine Park (black boundary) above the long-term average. (Map from Bureau of Meteorology)

As the global climate system warms, due to the increasing concentrations of atmospheric greenhouse gases, so do the tropical oceans. El Niño Southern Oscillation (ENSO) events are the major source of natural, inter-annual climate variability with significant impacts on Australia's

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[5] https://coralreefwatch.noaa.gov/satellite/analyses_guidance/global_coral_bleaching_2014-17_status.php

[6] http://www.gbrmpa.gov.au/media-room/latest-news/coral-bleaching/2017/second-wave-of-mass-bleaching-unfolding-on-great-barrier-reef. Surveys of this repetitive mass coral bleaching event were underway at the time of writing.

tropical terrestrial and marine environments. El Niño events are typically associated with a weaker summer monsoon and unusually warm late summer water temperatures along the Great Barrier Reef (Lough, 2007). This is the time of year when tropical corals are most at risk of reaching their thermal bleaching threshold due to unusually warm sea surface temperatures and may bleach. Conversely, during La Niña events, north-east Australia tends to experience above average rainfall and tropical cyclone activity (e.g. 2010-2011 in Figure 2).

Three mass coral bleaching events have been documented for the Great Barrier Reef in the past 20 years (in 1998, 2002 and 2016). The severity of each subsequent mass bleaching event has increased, with the 2016 event the most severe on record (GBRMPA, 2016a; Hughes et al., 2017). The 1998 and 2016 mass coral bleaching events were associated with El Niño events, and the magnitude of Great Barrier Reef warming was compounded by the increase in baseline water temperatures (as a result of global warming; Hughes et al., 2017).

1.5 Climate variation, trends and projections for the Great Barrier Reef region

Global and regional climate has varied and changed on a range of timescales in the past. It is now clear, however, that human activities, by increasing the concentrations of greenhouse gases in the atmosphere, are significantly and rapidly changing global and regional climate (Stocker et al., 2013). The concentration of the main greenhouse gas (carbon dioxide) is now 44% higher than pre-development levels and is likely to remain above 400 ppm for the foreseeable future (WMO, 2016), the highest levels in at least the past 800,000 years (Masson-Delmotte et al., 2013). As the global climate system warms so do the tropical oceans (Lough, 2012) and this has consequences, some of them already observable, for the Great Barrier Reef. In addition to regional warming, surface ocean climate of the Great Barrier Reef and coastal Queensland river catchments is modulated by ENSO events. El Niño events are typically associated with a weaker summer monsoon, fewer tropical cyclones, less rainfall and warmer late summer sea surface temperature, as observed in 2016. La Niña events are typically associated with a more active summer monsoon, more tropical cyclones and higher rainfall, as observed in 2011 (Lough, 2007). Within-season rainfall variability is also modulated by the Madden-Julian Oscillation and, over decadal timescales, by the Pacific Decadal Oscillation (Risbey et al., 2009). The combined impact of these different drivers is to introduce more variability into regional climate which, at times, can reinforce the intensity of extreme events. The magnitude of surface climate anomalies associated with ENSO is, for example, now magnified due to warming of the global climate system (e.g. Power et al., 2017).

1.5.1 Observed climate

Sea surface temperatures of the Great Barrier Reef have significantly warmed since the late 19th century (Figure 6).[7] Over the period 1880-2012, annual global land and sea temperatures increased by 0.85°C, with a marked increase of 0.72°C between 1951 and 2012 (Stocker et al., 2013). A slightly smaller increase was observed in the Great Barrier Reef, with annual sea surface temperatures warming by 0.74°C between 1880 and 2012 and by 0.63°C between 1951 and 2012. Updating through 2016 (the warmest year globally on record, www.ncdc.noaa.gov/sotc/global/201613), global temperatures have warmed 0.90°C[8] and the Great Barrier Reef by 0.80°C since 1880, and the rate of

[7] Monthly sea surface temperatures, January 1880 through December 2016, were extracted from the HadISST1 dataset (Rayner et al., 2003) for the 40 1° latitude by longitude boxes encompassing the Great Barrier Reef between 10.5°S and 24.5°S. Data are freely available from the UK Meteorological Office Hadley Centre: www.metoffice.gov.uk/hadobs/hadisst/data/download.html.

[8] Annual global land and sea temperatures, 1850–2016, were obtained from the HadCRUT4 dataset (Morice et al., 2012).

warming has accelerated since the 1950s (Table 1). March, April and May sea surface temperatures in 2016 on the Great Barrier Reef were the warmest months on record since 1880.

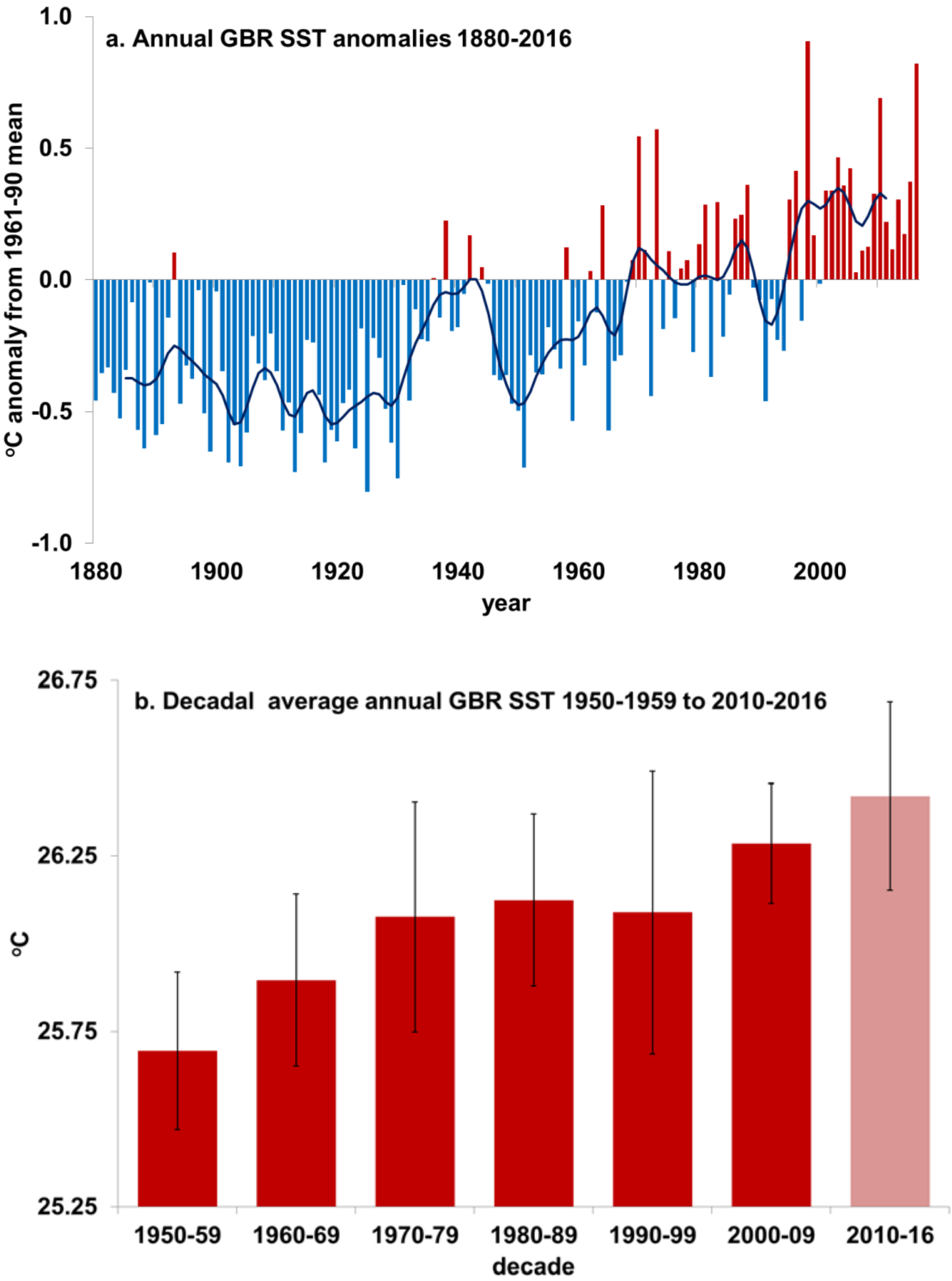

Figure 6: a) Annual Great Barrier Reef anomalies from 1961-1990 mean, 1880-2016; solid line is 10-year Gaussian filter to emphasise decadal variability, and b) decadal average annual Great Barrier Reef sea surface temperatures ± 1sd, 1950-1959 to 2010-2016. Note: most recent period is based on seven years.

Queensland rainfall, and resulting river flows into the Great Barrier Reef, is highly seasonal and highly variable from year to year and on decadal timescales. This is evident in the instrumental record of Queensland northern wet season (October–April) rainfall from the Bureau of Meteorology, 1901 through 2016[9], which shows, however, no significant overall trend in Queensland wet season rainfall over the period 1901-–2016.

Table 1: Decadal rates of warming for annual global land and sea temperatures and annual Great Barrier Reef sea surface temperature

Period	Global annual land and sea temperature warming	Great Barrier Reef annual sea surface temperature warming
1880-2016	0.07°C/decade	0.06°C/decade
1951-2016	0.12°C/decade	0.11°C/decade

As the climate changes, 'the environment in which all weather events occur is not what it used to be' (Trenberth et al., 2015) and warmer sea surface temperatures can increase the magnitude of extreme weather events. Long-term warming of the ocean around Australia, for example, increased the likelihood of extreme rainfall during the 2010-2011 La Niña event, resulting in the record rainfall in north-east Queensland in early 2011 (Ummenhofer et al., 2015).

As the warmer oceans expand and land-based ice melts, global and regional sea level is rising and is now ~19 cm higher than in the late 19th century (Church et al., 2013). A recent assessment (White et al., 2014) shows that, after allowing for the influence of ENSO events, average Australian sea level is rising at a similar rate to the global average at rates of 1.6 ± 0.2 mm/yr from 1966 to 2010 with a recent acceleration to 2.2 ± 0.5 mm/yr from 1990 to 2010. For Townsville, in the central Great Barrier Reef region, sea level rose by 1.3 mm/yr between 1966 and 2010. Given the similarity between recent Australian sea level trends and global averages, it is thus highly likely that further 21st century rises will also be similar.

1.5.2 Paleoclimate perspectives

High-resolution climate reconstructions[10] from corals and tree rings can provide a longer term perspective on Australian, Queensland and Great Barrier Reef climate variability and recent changes. A reconstruction of Burdekin River flow, based on coral luminescence in massive coral cores, for the period 1648-2011 shows that large flow events have increased in frequency from once in every 20 years (prior to European settlement of north-east Queensland between 1748 and 1847) to once every six years (between 1948 and 2011; Lough et al., 2015). Three of the four largest river flow events since 1648 have occurred in the past 43 years: 1974, 1991 and 2011. The reconstruction also showed a shift in the late 19th century to larger and more variable flows, which resulted in more frequent freshwater impacts on mid-shelf reefs. This climate shift, which would exacerbate sediment loads entering the Great Barrier Reef, appears to be linked with greater ENSO variability and rapid warming of the south-west Pacific, evident in independent paleoclimate records (Lough et al., 2015). Additionally, a new reconstruction of the eastern Australia summer Palmer Drought Severity Index[11], which is based on tree rings and coral records, shows that 2011 was likely the wettest summer in coastal Queensland in more than 500 years (between 1500 and 2012; Cook et al., 2016). A sea surface temperatures reconstruction from coral records for the western tropical Pacific (between

[9] www.bom.gov.au/climate/change/index.shtml#tabs=Tracker&tracker=timeseries

[10] Climate reconstructions are developed from paleoclimate archives, such as corals and trees, by calibrating the proxy record with instrumental climate data and using this regression to 'reconstruct' climate conditions prior to observational records.

[11] The Palmer Drought Severity Index is a measure of drought based on both rainfall amount and temperature.

1617 and 2006) suggests the Great Barrier Reef experienced its coolest temperatures in the past 400 years in the 1830s (possibly due to volcanic activity) followed by gradual warming and then accelerated warming through the 20th century (Tierney et al., 2015). A multi-proxy reconstruction of Australasian land and ocean temperatures (from 1000 to 2001), clearly demonstrates the unprecedented nature of recent rapid warming in the region and that post-1950 warming cannot be explained by natural variability alone—it can be attributed to anthropogenic forcing (the change in the planet's energy balance due to human influences, that is, warming caused by increased atmospheric greenhouse gas concentrations; Gergis et al., 2016).

1.5.3 Projected climate

Regional projections for natural resource management regions have recently been developed for Australia based on the most recent Intergovernmental Panel on Climate Change 5th Assessment Report Coupled Model Intercomparison Project Phase 5 (AR-5 IPCC CMIP5; cmip-pcmdi-llnl.gov/cmip5/) models for Representative Concentration Pathways[12] (RCPs: RCP2.6, RCP4.5, RCP6.0 and RCP8.5; CSIRO and Bureau of Meteorology, 2015). Projections are summarised here for the Monsoonal North, Wet Tropics and Marine and Coastal regions. It should be acknowledged that natural annual and decadal climate variability may obscure or enhance any long-term trends, particularly for rainfall.

With very high confidence[13] air temperatures are projected to warm in all seasons with more hot days and warm spells. There is also very high confidence that sea surface temperatures around Australia will continue to warm. The magnitude of projected warming is similar across Representative Concentration Pathways for the short term (2030) being approximately +0.6 to +0.8°C (relative to 1986-2005 average values) off Townsville. By the end of the 21st century (2090), warming could be as high as +1.3°C for RCP4.5 and +2.6°C for RCP8.5. Although changes in average rainfall are possible, there is no clear consensus among climate models on the direction and magnitude of change; more conclusive projections require better modelling of the spatial patterns of sea surface temperature change, especially in the tropical Pacific, which control convective activity influencing north-eastern Australian rainfall (Brown et al., 2016). There is, however, high confidence that the intensity of extreme rainfall events will increase. There is medium confidence that although there may be fewer tropical cyclones, those that do occur are likely to be more intense. There is very high confidence that sea level will continue to rise. The magnitude of sea level rise is similar across Representative Concentration Pathways for the short term (2030) being approximately 0.13m (relative to 1986-2005) for Townsville. The disparity between the different Representative Concentration Pathways increases over the 21st century, ranging from median increases of approximately 0.21–0.26 m by 2050, 0.30–0.43 m by 2070 and 0.38–0.64 m by 2090.

ENSO events have a significant impact on surface climate variability of the Great Barrier Reef and are likely to remain the dominant mode of natural inter-annual climate variability over the 21st century (Stocker et al., 2013). Although IPCC AR5 was somewhat equivocal as to how ENSO events might change in a warmer world, recent modelling and theoretical studies indicate that extreme El Niño

[12] Representative Concentration Pathways represent plausible ranges of radiative forcing of the climate system by different greenhouse gas emissions trajectories. RCP2.6 = low radiative forcing as a result of drastic mitigation strategies; RCP4.5 and 6.0 = intermediate radiative forcing with some level of mitigation; RCP8.5 = high radiative forcing with little curbing of greenhouse gas emissions (the trajectory currently being tracked).

[13] In describing climate projections for the Great Barrier Reef, the IPCC confidence assessment ratings are used; these are *very low*, *low*, *medium*, *high* and *very high* confidence based on evidence and agreement among climate models (see Box TS.1 in Stocker et al., 2013).

events (e.g. 1997-1998, 2015-2016) and extreme La Niña events (e.g. 2010-2011) are both likely to occur more frequently as the world continues to warm (Cai et al., 2015).

2. Water quality in the coastal and marine areas of the Great Barrier Reef region

2.1 Exposure of coastal and marine environments to land-based run-off

Land-based run-off containing suspended sediment, nutrients and photosystem II-inhibiting (PSII) pesticides is exported to the Great Barrier Reef, with most of these pollutants being delivered during river floods (Brodie et al., 2012). Exposure of the Great Barrier Reef to land-based run-off has been examined using physical oceanographic studies of river plumes, modelling studies and satellite observations. Recent studies have started to integrate these areas of research, including *in situ* water quality data, significantly improving our understanding of potential exposure of, and impacts on, reef ecosystems in the Great Barrier Reef (Devlin et al., 2013a; Brinkman et al., 2014; Schiller et al., 2014; Delandmeter et al., 2015; Devlin et al., 2015a; Jones, E.M. et al., 2016). Since the last consensus statement, the eReefs project has progressed significantly to develop a modelling, hindcasting and forecasting capability for the ocean circulation and water quality on the Great Barrier Reef (Schiller et al., 2014). The models are calibrated, validated and forced with available biophysical observations from satellites (e.g. Chlorophyll-*a*), tide gauges (sea surface height), moorings (e.g. water movement) and ocean glider data (e.g. temperature and salinity) (Schiller et al., 2014; Baird et al., 2016; Jones, E.M. et al., 2016). Validation with *in situ* observations of water quality (e.g. Chlorophyll-*a*) is currently limited to inshore and coastal Great Barrier Reef Marine Monitoring Program sites (Lønborg et al., 2016), and monitoring needs to be extended to other areas (e.g. Cape York, mid- and offshore areas) for improved calibration of the eReefs biochemical component.

Satellite observations have been used to map the extent of river flood plumes and qualitatively assess the dispersal of pollutants in these plumes (Álvarez-Romero et al., 2013; Jones and Berkelmans, 2014; Devlin et al., 2015a), as well as to detect, map and analyse water quality parameters in coastal and marine waters (Devlin et al., 2013a; Blondeau-Patissier et al., 2014; Petus et al., 2014a; Baird et al., 2016; Hedley et al., 2016; Jones, E.M. et al., 2016; Petus et al., 2016). Recent work has confirmed that flood plumes following heavy rainfall events can extend long distances and cover large areas (Devlin et al., 2012; Butler et al., 2013; Devlin et al., 2013a; Jones and Berkelmans, 2014; Petus et al., 2014a; Delandmeter et al., 2015; Devlin et al., 2015a), such as the plume extending for 70 km^2 and covering an area of ~200 km^2 following the 2010-2011 Fitzroy River flood (Jones and Berkelmans, 2014). The extent of river flood plumes, and consequently the exposure of coral reefs and seagrass meadows to flood plumes and associated land-based pollutants (e.g. suspended sediment, dissolved inorganic nitrogen) shows strong inter-annual variation (Devlin et al., 2012; Álvarez-Romero et al., 2013; Devlin et al., 2015a). This variability in timing and intensity of floods influences the ecological responses in the Great Barrier Reef lagoon, such as the spatial distribution and temporal dynamics of water clarity (Fabricius et al., 2014; Fabricius et al., 2016), phytoplankton biomass (Devlin et al., 2013a) and marine microbial communities (Angly et al., 2016). By combining remote sensing–derived exposure maps with *in situ* water quality data, the likelihood and magnitude of the river plume risk to coral reefs and seagrass meadows can now be assessed across large regions and multi-annual periods (Petus et al., 2014a; Lønborg et al., 2016; Petus et al., 2016).

Increased exposure to land-based sediment associated with river run-off events since European settlement has been confirmed using cores of reef sediment (Lewis et al., 2014) and corals (McCulloch et al., 2003; Lewis et al., 2007, 2012). Biogeochemical records from coral cores collected at Dunk Island in the Wet Tropics region demonstrate an eightfold increase in phosphorus levels between 1949 and 2008, associated with increases in fertiliser applications and riverine run-off of particulate phosphorus (Mallela et al., 2013). Jupiter et al. (2008) show a strong positive correlation between nitrogen stable isotopes in coral cores (i.e. coral skeletal $\delta^{15}N$, a tracer of nitrogen) at two inshore reefs and major flow of the Pioneer River, during the period 1968-2002. In contrast, Erler et al. (2016) found an unexpectedly stable record of $\delta^{15}N$ in a coral core from Magnetic Island in the central Great Barrier Reef over a 168-year period (1820-1987), using a new analytical technique. This result suggests that regional nitrogen cycling (as a proxy for nutrients) did not significantly change over this time period but the robustness of this preliminary result, based on a single coral, is currently being tested based on multiple coral records.

Coral luminescence series from 20 *Porites* coral cores also show that rainfall variability has increased along a large part of the Great Barrier Reef since the late 1800s, with wet and dry periods becoming increasingly extreme (Lough, 2011). A recent reconstruction of the Burdekin River flow (1648-2011) using luminescent lines in corals highlighted that the frequency of high flow events has increased in the Great Barrier Reef, resulting in more frequent freshwater impacts on mid-shelf reefs since the mid-1900s (Lough et al., 2015). The alteration of land use during and after European settlement has resulted in increased sediment loads being delivered to the Great Barrier Reef (see Chapter 2). This has been further exacerbated by the increased frequency in flows brought about by climate change (Lough et al., 2015). Results from sediment cores show that most fine sediment (<63µm) delivered from the Burdekin River is retained within 50km of the Burdekin River mouth (Lewis et al., 2014). These results were supported by subsequent hydrodynamic modelling of the transport and fate of riverine fine sediment (<15.6µm) exported from the Burdekin River (Delandmeter et al., 2015). Differences in the seasonal cycles in spectral luminescence ratios (green/blue) between a mid-Holocene (4500 years ago) and present-day coral are consistent with higher levels of vegetation within river catchments during the mid-Holocene (Roche et al., 2014). However, Lough and Cantin (2014) suggest, based on replicate luminescence records in coral samples, that the north-eastern Australian summer monsoon was weaker than present (with substantially lower Burdekin River flow) 6000 years ago, which matches the change in orbital forcing affecting the monsoon systems at that time.

2.2 Spatial and temporal distribution of pollutants and associated indicators

2.2.1 Nutrients/Chlorophyll-*a*

Riverine discharges of nitrogen and phosphorus have increased following agricultural development of the Great Barrier Reef catchment (Chapter 2) and contributed to elevated Chlorophyll-*a*[14] and nutrient concentrations in receiving waters (Brodie et al., 2011). Concentrations of different nutrient constituents as well as Chlorophyll-*a* are monitored in the Great Barrier Reef Marine Monitoring Program (Lønborg et al., 2016) and compared with water quality guideline trigger values (GBRMPA, 2010). However, the current water quality metric used in the Reef Report Card (Australian and Queensland governments, 2016) is based on remote sensing–derived data only and does not include *in situ* collected data. Recent research showed that estimates-based satellite observations in the Great Barrier Reef for concentrations of Chlorophyll-*a* and total suspended sediment (TSS) have a degree of uncertainty in some geographic areas, particularly where waters are shallow and naturally turbid, such as Princess Charlotte Bay and Shoalwater Bay, and under certain circumstances including wet season conditions, where there is less satellite imagery available due to cloud cover and algorithms to derive estimates of Chlorophyll-*a* are less reliable (Blondeau-Patissier et al., 2014; Jones, E.M. et al., 2015; Waterhouse and Brodie, 2015; Baird et al., 2016; Jones, E.M. et al., 2016). An *in situ* monitoring network for the calibration and validation of remote sensing estimates and water quality models remains important. Following a review of the approach by the Marine Monitoring Program providers in 2016 (e.g. Tracey et al., 2016), a new approach for an improved water quality

[14] Chlorophyll-*a* concentration is a measure of phytoplankton biomass in a water body; in coastal waters, it can reflect changes in river nutrient loads. In the Great Barrier Reef Water Quality Guidelines, chlorophyll-*a* is used as a proxy for dissolved inorganic nitrogen (GBRMPA, 2010), generally considered the most readily available nitrogen form for uptake by marine plants such as phytoplankton.

metric for the Reef Report Card is currently being developed as part of a National Environmental Science Programme's Tropical Water Quality Hub research project.[15]

Both remote sensing and *in situ* water quality data show that patterns of Chlorophyll-*a* distribution in the Great Barrier Reef lagoon are complex and highly heterogeneous spatially and over seasonal and inter-annual cycles, and Chlorophyll-*a* concentrations are typically elevated during the wet season (Devlin et al., 2013a; Jones, E.M. et al., 2015; Oke et al., 2015; Jones, E.M. et al., 2016). The strong spatial and temporal variability in Chlorophyll-*a* concentrations contrasts with water quality trigger values that reflect cross-shelf gradients (i.e. higher inshore than offshore) but not any latitudinal differences (GBRMPA, 2010), and continues to be a point of discussion in the scientific literature (Bell et al., 2014a; Bell et al., 2014b; Furnas et al., 2014). Hence, understanding the natural patterns of across the Great Barrier Reef is a key knowledge gap that requires resolution, including for the purpose of refining existing water quality trigger values.

Of the different nutrient constituents, dissolved inorganic nitrogen is thought to pose the largest risk to the Great Barrier Reef ecosystems (Brodie et al., 2015) as it is readily available for uptake by marine plants such as phytoplankton, macroalgae and algal symbionts in corals. The uptake by phytoplankton in particular contributes to long-distance transport of nitrogen as it goes through repeated cycles of growth, decay and remineralisation (D'Angelo and Wiedenmann, 2014). The largest amount of terrestrial nitrogen, however, is discharged in particulate form (i.e. particulate nitrogen; Chapter 2). It is thought that most of the particulate nitrogen is deposited near river mouths or incorporated into suspended organic aggregates (Brodie et al., 2015). The relative bioavailability of terrestrially derived particulate nitrogen in the marine environment, however, is still unclear and deserves further study, given that particulate nitrogen is the largest constituent of terrestrially derived nitrogen.

Across the Great Barrier Reef lagoon, concentrations of dissolved nutrients (dissolved inorganic nitrogen, dissolved organic nitrogen, dissolved inorganic phosphorus, dissolved organic phosphorus, dissolved organic carbon, Silica), particulate nutrients (particulate nitrogen, particulate phosphorus, particulate carbon) as well as TSS and Chlorophyll-*a* are generally more variable during the wet than the dry season (McKinnon et al., 2013; Lønborg et al., 2016). Cross-shelf location had the largest effect on water quality, with higher values recorded inshore, which is also reflected in patterns of pelagic metabolism (McKinnon et al., 2013). A cross-shelf study showed that numbers and production of plankton groups, including virioplankton, bacterioplankton and phytoplankton, correlated with elevated levels of particulate carbon, nitrogen and phosphorus (Alongi et al., 2015). This study showed a tight coupling of pelagic and benthic shelf processes, with phytoplankton deriving around half of their essential nutrients (nitrogen, phosphorus) from nutrient regeneration in the sediments and turbulent resuspension of the sediments (Alongi et al., 2015). This suggests that particulate nitrogen may be more important in driving phytoplankton production (and hence Chlorophyll-*a* levels) than is currently appreciated, further emphasising the need to determine its relative bioavailability given that particulate nitrogen is the largest constituent of terrestrially derived nitrogen.

2.2.2 Sediment/Turbidity

Riverine discharges of suspended sediment and particulate nutrients have increased following agricultural development of the Great Barrier Reef catchments (Chapter 2) and have affected water clarity in Great Barrier Reef receiving waters (Bartley et al., 2014; Fabricius et al., 2014; Lewis et al.,

[15] nesptropical.edu.au/index.php/round-3-projects/project-3-2-5/

2015a; Lewis et al., 2015b; Fabricius et al., 2016). Water clarity indicators, namely turbidity[16], Secchi depth and TSS, are monitored in the Great Barrier Reef Marine Monitoring Program (e.g. Lønborg et al., 2016) and compared with water quality guideline threshold values (GBRMPA, 2010). However, as described above, the current water quality metric used in the Reef Report Card is based on remote sensing–derived data only (see '2.2.1 Nutrients/Chlorophyll-*a*'). Time-series measurements for turbidity in coastal waters of the Great Barrier Reef confirm strong spatial and temporal variability at individual reefs (Browne et al., 2013), and within and across regions (Macdonald et al., 2013). Notwithstanding this high variability, a study using 11 years of daily river load, oceanographic and remote sensing data showed that water clarity was related to river loads after statistical removal of wave and tidal effects (Fabricius et al., 2014; Fabricius et al., 2016). Averaged across the Great Barrier Reef, river loads reduced water clarity by 50%, and recovery took approximately six months. The river load effects were strongest in the Cairns region (where primary outbreaks of crown-of-thorns starfish originate), where rivers affected water clarity even on outer shelf reefs.

Discharge from the Burdekin and Fitzroy rivers and associated particulate constituents influence turbidity and photic depth of adjacent inshore waters (Lewis et al., 2015a; Lewis et al., 2015b). Only a small proportion of the sediment that has been eroded from the Burdekin catchment is transported to the mid and outer reefs (Bartley et al., 2014; Lewis et al., 2014; Bainbridge et al., 2016). Most fine sediment (<63µm) delivered from the Burdekin River is retained within 50km of the Burdekin River mouth (Delandmeter et al., 2015; Lewis et al., 2014), but wind- and tide-driven resuspension can result in remobilisation of this fraction (Bartley et al., 2014). The sediment fraction transported in the flood plumes >1 km offshore is generally the clay to fine silt (<4–16µm) fraction (Bainbridge et al., 2012; Bartley et al., 2014; Delandmeter et al., 2015; Bainbridge et al., 2016). This fraction, which also contains particulate nitrogen and phosphorus (Lewis et al., 2015a), can form organic-rich flocs and result in increased turbidity in coastal areas following major discharge events from the Burdekin River (Fabricius et al., 2014; Lewis et al., 2014, Lewis et al., 2015a).

Since the last Scientific Consensus Statement (Brodie et al., 2013), capital and maintenance dredging for Great Barrier Reef ports has received local, national (Lankester et al., 2015; McCook et al., 2015), and global attention, for example by the UNESCO.[17] Major expansions were planned for Cairns, Townsville, Abbott Point, Hay Point and Gladstone, with capital dredge material to be placed in the Great Barrier Reef Marine Park (Brodie, 2014; McCook et al., 2015). The Gladstone expansion has since been completed (Brodie, 2014); however, if expansions are to proceed at the other ports the capital dredge spoil must now be disposed of on land (McCook et al., 2015). Dredge material derived from maintenance dredging can still be deposited at sea in the Great Barrier Reef World Heritage Area. The turbidity implications of dredging activities in reef environments are highly complex over time frames from hours to weeks (Jones, R. et al., 2015a). The direct and indirect effects of dredging are severe within the dredging footprint and could be significant at local and regional scales (McCook et al., 2015). The potential impact of maintenance dredging is currently the topic of a National Environmental Science Programme's Tropical Water Quality Hub research project.[18]

2.2.3 Pesticides

Compared with those for sediment and nutrients, fewer monitoring and research projects have been conducted into the occurrence of pesticides in the Great Barrier Reef coastal and marine

[16] Turbidity is a measure of light scattering caused by fine suspended particles, such as clay and silt, detritus, microbes and phyto- and zooplankton. Turbidity is affected by a wide range of factors, including natural ones such as wind, waves and currents, as well as anthropogenic ones such as dredging and increased land-based run-off.

[17] See whc.unesco.org/archive/2015/whc15-39com-7BAdd-en.pdf

[18] See nesptropical.edu.au/index.php/round-2-projects/project-2-1-9/

environments. Devlin et al. (2015b) provide a recent review on the transport and fate of pesticides in the Great Barrier Reef. This review, and additional monitoring (Gallen et al., 2015) since then, found that a wide range of PSII herbicides and other pesticides (e.g. terbutryn, imidacloprid, imazapic and metolachlor) are detectable in the Great Barrier Reef environment. Flood plume monitoring shows that PSII herbicides are generally 'conservatively mixed', that is, concentrations become increasingly diluted as the freshwater discharge progressively mixes with seawater (Devlin et al., 2015b). In 2013-2014, pesticide concentrations in the Great Barrier Reef marine environment did not exceed Great Barrier Reef water quality trigger values (GBRMPA, 2010; Gallen et al., 2015). These trigger values, however, are mostly of low or moderate reliability, and refinement of current and new water quality trigger values for agricultural pesticides is the topic of current National Environmental Science Programme's Tropical Water Quality Hub research.[19] Herbicides commonly detected in the Great Barrier Reef show long persistence in seawater simulation experiments with native bacterial populations and coastal seawater from the Great Barrier Reef (Mercurio et al., 2014; Mercurio et al., 2015). Half-lives of PSII herbicides ametryn, atrazine, diuron, hexazinone and tebuthiuron are all greater than a year (Mercurio et al., 2015), indicating high persistence and explaining their year-round presence in the Great Barrier Reef (Gallen et al., 2015). Given that herbicide persistence is higher in low-light conditions, it is likely that limited degradation would occur during transport in wet season flood plumes (Mercurio et al., 2014; Mercurio et al., 2015). The degradation of diuron may be mediated by microbes in the Great Barrier Reef inshore.

2.2.4 Ocean acidification

Ocean acidification[20] is increasingly recognised as an important water quality pressure on the Great Barrier Reef and on coral reefs globally. The levels of pCO_2 on Great Barrier Reef inshore reefs have disproportionately increased compared to atmospheric levels, and inshore waters now have an environment that may adversely affect coral calcification and is more beneficial for benthic algae, seagrasses and phytoplankton (Uthicke et al., 2014). In the Great Barrier Reef lagoon, regular monitoring of carbonate chemistry to document ocean acidification is still in development[21], and broadscale spatial and temporal patterns are only starting to be recognised. An analysis of a multi-year dataset of carbonate chemistry parameters (aragonite saturation, pCO_2) collected at the water quality monitoring sites of the Great Barrier Reef Marine Monitoring Program and some offshore reefs showed marked differences across the shelf. On inshore reefs, metabolism of the abundant organic matter releases additional CO_2 during the night, which reduces pH. A recent application of the eReefs model confirmed the cross-shelf patterns of ocean acidification parameters and showed that carbonate dissolution already exceeded net calcification in inter-reefal areas of the Cairns/Cooktown to Townsville Whitsunday Management Areas, but also highlighted that the many factors influencing ocean acidification on the Great Barrier Reef are still poorly understood (Mongin et al., 2016).

2.2.5 Other pollutants

For pollutants other than pesticides, few monitoring and research projects have been conducted since the 2013 Scientific Consensus Statement (Kroon et al., 2015a). In addition, existing monitoring information is not always made available to research scientists by the respective custodians (Kroon et al., 2015a). This means that not all existing environmental datasets can be included in, for

[19] nesptropical.edu.au/index.php/round-3-projects/project-3-1-5/

[20] Ocean acidification is the lowering of seawater pH, caused by the absorption of carbon dioxide (CO_2) by the oceans from the atmosphere (Uthicke et al., 2014).

[21] See www.barrierreef.org/science-with-impact/ship-of-opportunity

example, environmental risk assessments, with a potential flow-on effect on policy and management not being based on a complete picture of existing relevant information.

Antifouling paint components such as metal pigments and booster biocides, including tributyltin, have been detected in the coastal and marine environments of the Great Barrier Reef and are mainly sourced from shipping (Kroon et al., 2015a). Areas exposed to shipping include ports and marinas, port anchorage areas, shipping lanes, tourism moorings, shipping incidents and groundings and military exercise and range areas. In the Great Barrier Reef region, recent monitoring programs associated with dredging and port developments have demonstrated that tributyltin is still present in the water column and/or sediment in the harbours of Abbot Point, Cairns, Gladstone, Hay Point and Townsville (GHD Pty Ltd, 2005; Jones et al., 2005; DEHP, 2012; Port of Townsville Limited et al., 2013; Ports North, 2015; Port Curtis Integrated Monitoring Program Inc., 2016). Bioaccumulation of tributyltin in oysters has been reported in Rosslyn Bay (Keppel Island), Cairns Port and Gladstone Port (Mortimer, 2004). Enrichment of tributyltin in oysters and mud whelks has been documented in certain areas around Port Curtis (Jones et al., 2005), including concentrations that exceed those known to be linked to imposex[22] in gastropods (Jones et al., 2005). Leaching and release of antifouling components from large vessels will also occur while underway along shipping lanes and while at anchor at port anchorage areas, but no monitoring data for either sediment or water is available for this. Significant amounts are likely to be released into these environments based on a total copper leach flux of around 0.8–3.1 for an average bulk carrier for Great Barrier Reef coal ports (PGM Environment, 2012). Determining whether leaching of copper from antifouling paints at Great Barrier Reef port anchorage sites is an identifiable risk to the Great Barrier Reef values has been identified as an action in the North-East Shipping Management Plan (North-East Shipping Management Group, 2014). Ship groundings in the Great Barrier Reef have resulted in metal and tributyltin concentrations in water and sediments that are several orders of magnitudes higher than the ANZECC/ARMCANZ guidelines (Haynes et al., 2002; Haynes and Loong, 2002; Negri et al., 2002; GBRMPA, 2011).

Metals and metalloids[23] have been detected in water, sediment and biota in the Great Barrier Reef and Torres Strait marine ecosystems (Kroon et al., 2015a). Most, if not all recent monitoring programs with information on metals and metalloids have been conducted in and around the ports of Abbot Point, Cairns, Gladstone, Hay Point and Townsville. In these environments, the monitoring data indicate that dissolved metal/metalloid concentrations in Great Barrier Reef surface water were typically low, although some exceedances of water quality guidelines were recorded (for more detail, see Kroon et al., 2015a). Exceedances of low level sediment quality guidelines were more common in general, more common for more metals, and were observed for at least four out of the five main ports (Kroon et al., 2015a). Metals and metalloids were detected in a variety of marine biota from the Great Barrier Reef and Torres Strait marine ecosystems (see reviews in Berry et al., 2013; Kroon et al., 2015a), with some concentrations of metals exceeding generally expected levels for crustaceans and molluscs as defined by Food Standards Australia New Zealand (Food Standards Australia New Zealand, 2015). However, current levels of metal bioaccumulation in Great Barrier Reef organisms are unknown as there are no data available post 2009. Metals and metalloids can exert toxic (Trenfield et al., 2015; Trenfield et al., 2016) and sublethal effects (Reichelt-Brushett and Hudspith, 2016) at concentrations measured in Great Barrier Reef and Torres Strait marine ecosystems (for more detail, see Kroon et al., 2015a).

[22] Imposex is an endocrine disorder in marine gastropod molluscs caused by certain pollutants where female gastropods develop male sex organs.

[23] Chemical elements with properties intermediate between those of typical metals and non-metals.

Concentrations of **polycyclic aromatics hydrocarbons** in areas of the Great Barrier Reef away from human activity are generally low (Kroon et al., 2015a), ranging in sediment from undetectable at offshore sites to low at island locations and high in busy ports (Smith et al., 1985; Smith et al., 1987). Polycyclic aromatics hydrocarbons were also detected in sediments of four rivers that flow into the Great Barrier Reef lagoon (Humphrey et al., 2007) and in surface sediments and sediment traps from nearshore sites around Abbot Point/Mackay to offshore sites close to mid-shelf coral reefs (Burns and Brinkman, 2011). Results from monitoring programs associated with dredging and port developments have typically detected total polycyclic aromatics hydrocarbons concentrations below guideline levels (Kroon et al., 2015a). Ship incidents and groundings can result in the release of polycyclic aromatics hydrocarbons, such as the loss of ~4 t of fuel oil from the *Shen Neng 1* following the grounding on Douglas Shoal off Gladstone in 2010 (GBRMPA, 2011), and the loss of ~25 t of fuel oil from the *Global Peace* following a collision with a tugboat within Gladstone Harbour in 2006 (Andersen et al., 2008; Melville et al., 2009). With the exception of the *Global Peace* oil spill (Andersen et al., 2008; Melville et al., 2009), generally little monitoring of polycyclic aromatics hydrocarbons has been undertaken following large fuel spills from shipping in the Great Barrier Reef. Polycyclic aromatics hydrocarbons are often not detected in biota of contaminated sites, potentially because they can be rapidly metabolised (Kleinow et al., 1987). However, several field studies in the Great Barrier Reef region have identified polycyclic aromatics hydrocarbons or biomarkers for exposure in marine organisms, including fish, crabs, giant clams and sea cucumbers (Smith et al., 1984; Coates et al., 1986; Smith et al., 1987; Humphrey et al., 2007; Negri et al., 2009; van Oosterom et al., 2010).

Marine debris has been detected along the coastlines and in coastal and marine waters of the Great Barrier Reef (Haynes, 1997; Reisser et al., 2013; Hardesty et al., 2014; Hall et al., 2015; Tangaroa Blue Foundation, 2015). In a total of 1121 beach clean-ups conducted between January 2008 and October 2015, a total of 1,781,068 anthropogenic items were collected primarily consisting of those classified as plastics (n = 1,440,383; 81%; Kroon et al., 2015a). Fragmentation of marine plastics will ultimately contribute to microplastics[24] contamination (GESAMP, 2015), which has recently been documented in Great Barrier Reef waters (Hardesty et al., 2014; Hall et al., 2015). Both studies identified that microplastics fragments most likely derived from single-use disposable packaging and fishing equipment. Based on recent evidence from overseas (Wright et al., 2013; Gall and Thompson, 2015), the potential impacts of plastic pollution in the Great Barrier Reef are likely to be much more widespread, but are currently not quantified. Both entanglement in and ingestion of plastic debris have been reported for marine species in the Great Barrier Reef and Torres Strait marine ecosystems including turtles, cetaceans, dugong and seabirds (Ceccarelli, 2009). Physical injury by marine debris, such as derelict fishing lines, was found to increase the occurrence of coral disease on the Great Barrier Reef (Lamb et al., 2015; Lamb et al., 2016). Microorganisms and invertebrates have been detected on floating marine plastic in Great Barrier Reef waters (Reisser et al., 2014), indicating a new dispersal pathway for marine organisms. Only a few projects have specifically examined the sources of marine debris pollution in the Great Barrier Reef region (Haynes, 1997; Griffin, 2008; Hardesty et al., 2014). Most of the marine debris found on islands and cays in the Far Northern Management Area of the Great Barrier Reef is likely derived from oceanic and local shipping sources (Haynes, 1997), including lost and discarded fishing gear from commercial and recreational fishing vessels. The highest concentrations (40,000–80,000 pieces/km^2) of plastic particles were recorded between Shoalwater Bay and Townsville in February 2013 and were associated with large flooding

[24] Microplastics are generally defined as plastic particles <5 mm in diameter (Arthur et al., 2009), with the terms 'primary' and 'secondary' microplastics referring to particles being either specifically manufactured for particular applications or produced as a result of fragmentation from larger items.

events (Hardesty et al., 2014) due to ex-tropical cyclone Oswald (Bureau of Meteorology, 2014), thus most likely derived from land-based sources.

Pharmaceuticals and personal care products have been detected in wastewater treatment plants effluents discharging into rivers that flow into the Great Barrier Reef lagoon (O'Brien et al., 2014; Scott et al., 2014). These two studies provide information for just five personal care products and 26 pharmaceuticals, with the total number of pharmaceuticals and personal care products likely to be much higher based on Australian and overseas studies (Kroon et al., 2015a). In the majority of cases, pharmaceuticals measured in treated sewage were at generally low concentrations between 10 and 500ng/L, with some higher exceptions (e.g. up to 2.3µg/L for iopromide; O'Brien et al., 2014). In river water, paracetamol was reported in the Fitzroy region at 4.1µg/L (Scott et al., 2014). The personal care products detected in wastewater treatment plants effluent include acesulfame (an artificial sweetener), iopromide (an X-ray contrast agent) and triclosan (an antimicrobial; O'Brien et al., 2014). Many of these measurements come from an individual sampling event, so cannot be taken as representative concentrations. To our knowledge, no studies have been published that examine the potential toxicity or sublethal impacts of pharmaceuticals and personal care products on tropical marine organisms in the Great Barrier Reef. Overseas studies have shown that very low concentrations of sunscreen (10 µL/L) can result in complete bleaching of hard corals within 96 hours (Danovaro et al., 2008). In addition, numerous studies have demonstrated that sublethal outcomes from exposures to low concentrations of pharmaceuticals and personal care products can often have implications for higher level biological processes, for example survival, health and population fitness (Ankley et al., 2010; Groh et al., 2015).

Other pollutants detected and/or potentially present in Great Barrier Reef waters include coal particles, nanomaterials, perfluorooctane sulfonate and perfluorooctanoic acid. For coal particles, limited recent and publicly available data exist for the broader Great Barrier Reef marine ecosystems (Kroon et al., 2015a). For example, varying sizes of coal particles were found in sediment samples in the proximity of loading facilities at Abbot Point (cited as: WBM, 2005 in Toki et al., 2012). Hydrocarbon markers for coal (Burns, 2014) have been identified in sediments and sediment traps in the Great Barrier Reef lagoon, with the highest concentrations detected closer to the coast (Burns and Brinkman, 2011). Controlled laboratory exposures of three taxa commonly found within the Great Barrier Reef (the hard coral *Acropora tenuis*, the reef fish *Acanthochromis polyacanthus* and the seagrass *Halodule uninervis*) have demonstrated sublethal and lethal effects (Berry et al., 2016; Berry et al., 2017). However, the lack of available monitoring data in the Great Barrier Reef precludes comparing these effects in the laboratory to potential impacts in the field. Nanomaterials have not been monitored in the Great Barrier Reef coastal and marine ecosystems, nor has their potential fate or effect on Great Barrier Reef organisms been examined. Potential effects of nanomaterial exposure in marine organisms have been demonstrated elsewhere (Miao et al., 2010; Miller et al., 2010; Wong et al., 2010; Buffet et al., 2011; Matranga and Corsi, 2012; Baker et al., 2014; Jovanovic and Guzman, 2014; Sanchez-Quiles and Tovar-Sanchez, 2014; Suwa et al., 2014). Specifically, the direct release of nanomaterials in consumer products into the marine environment has raised concerns relating to their potential effects on coral and phytoplankton (Miao et al., 2010; Miller et al., 2010; Suwa et al., 2014). Finally, perfluorooctane sulfonate and perfluorooctanoic acid have recently been detected in groundwater, surface water and sediment at the RAAF Base Townsville (GHD Pty Ltd, 2016). Concentrations at several locations for groundwater and surface water exceeded screening levels for drinking water (GHD Pty Ltd, 2016). As of February 2016, fluorine-based foams were listed as being used at the airports of Cairns, Hamilton Island, Mackay, Rockhampton and Townsville, with historical use listed for Bundaberg and Proserpine (Airservices Australia, 2016). Perfluorooctane sulfonate and perfluorooctanoic acid have not been monitored in the Great Barrier Reef coastal and marine ecosystems, nor has their potential fate or effect on Great Barrier Reef organisms been examined.

3. Condition of coastal freshwater wetlands in the Great Barrier Reef region

3.1 Extent and condition of coastal freshwater wetlands

Freshwater wetlands in the Great Barrier Reef catchments can provide a vital role in coastal and marine ecosystems by retaining sediment, absorbing and transforming pollutants, slowing overland flow and providing nurseries for freshwater and marine species (DEHP, 2016). The ecological health and the economic value of the Great Barrier Reef depend in part on the health of its wetlands (Stoeckl et al., 2011; Sheaves et al., 2014). In the catchments of the Great Barrier Reef, there are over 15,000km^2 of wetlands (artificial/highly modified, lacustrine, palustrine, riverine and estuarine), many of which have been extensively modified or lost over the last 100 years (Wetland*Info*, 2016). The total extent of near-natural wetlands remaining in Great Barrier Reef catchments in 2013[25] equates to 85% of the pre-European extent (Australian and Queensland governments, 2015). Of the 737,500ha of near-natural wetlands (non-riverine and including estuarine) mapped in the Great Barrier Reef catchment in 2013, palustrine wetlands (e.g. vegetated swamps) comprised 281,600ha and lacustrine wetlands (e.g. lakes) comprised 22,400ha (DPC, 2014). Rates of wetland loss between 2009 and 2013 were lower in most regions than for previous periods (<0.1%). However, the condition of wetlands in the Great Barrier Reef catchment has declined considerably (GBRMPA, 2014a), and many wetlands are under high threat of degradation (Brodie and Mitchell, 2005).

Coastal freshwater wetlands have suffered a range of cumulative pressures such as increased nutrient and sediment loads, loss of connectivity and changes in hydrology (Kroon et al*.,* 2013; Tsatsaros et al*.,* 2013; GBRMPA, 2016b). Catchment modifications have led to secondary impacts such as weed infestations, low water quality and algal blooms (GBRMPA, 2016b). For example, wetlands south of Cairns are characterised by extensive development, particularly on coastal lowlands (GBRMPA, 2014a). Freshwater wetlands in catchments with less intensive development are usually in better condition; however, they are still considered highly vulnerable due to impacts such as grazing, soil erosion, feral animals, weeds, altered fire regimes and climate change (GBRMPA, 2014a). Wetlands within the Great Barrier Reef region vary greatly among catchments, thus their response to impacts may differ. Some of the key impacts on freshwater wetlands will be discussed in the following sections.

3.2 Water quality and other pressures affecting coastal freshwater wetland condition

3.2.1 Nutrients

Water entering freshwater wetlands within the Great Barrier Reef catchment is affected by high nutrient inputs (Brodie and Mitchell 2005; Tsatsaros et al., 2013; Davis et al., 2017). Nutrients such as nitrogen and phosphorus occur naturally in freshwater wetlands (Arthington et al., 1997); however, annual discharge of nutrients has more than doubled since European settlement (see Chapter 2). The increase of NO_X loads (nitrite + nitrate) in rivers follows a longitudinal gradient with concentrations increasing as water travels downstream (Connolly et al., 2015).

Increased nutrient loads to wetlands can lead to sudden, and sometimes irreversible, changes in factors such as increases in productivity, shifts in species dominance and composition, changes in carbon and nutrient cycling, disruption of balance between respiration and photosynthesis and modified trophic interactions (Wetzel, 2001; Verhoeven et al., 2006). Excess nutrients can lead to excessive algal growth, which can affect invertebrate fauna by reducing light and suitable substrate (Connolly and Pearson*,* 2007). Nutrients coupled with high temperatures can cause declines in

[25] Only data to 2013 were available at the time of writing.

macroinvertebrate and fish (Mantyka-Pringle et al., 2014) and the overgrowth of weeds (Brodie and Mitchell, 2005).

The effects of nutrients on freshwater wetlands in the Great Barrier Reef catchment are largely driven by rainfall. During wet seasons, large amounts of nutrients, sediments and pesticides can be rapidly mobilised and transported downstream (Brodie et al., 2012; Joo et al., 2012; Davis et al., 2017). During the dry season, poor water quality can be exacerbated by low flushing and high organic matter inputs from dying organisms (Burford et al., 2008). For example, in the Burdekin River, algal blooms and poor water quality are common in the dry season during periods of low flows (Congdon and Lukacs, 1996; Davis et al., 2017). Future changes in the frequency, timing and intensity of rainfall in the region (CSIRO and BoM, 2015) will have consequences on the impact of nutrients in freshwater wetlands (GBRMPA, 2016b).

3.2.2 Sediments

Excess sediment can put pressure on freshwater wetlands directly through increases in turbidity, but also indirectly, as sediments carry pollutants and nutrients attached to them (Neil et al., 2002; Bainbridge et al., 2009). The size, shape and chemical characteristics of sediments influence their impact on wetlands (Wood and Armitage, 1997). The impacts of increased sediments in naturally clear systems (e.g. in upland Wet Tropics streams) might be greater compared to the impacts on naturally turbid catchments (e.g. lowland dry tropics rivers; Connolly and Pearson, 2007; Davis et al., 2017).

A significant impact of excess sediments can also be the infilling of some wetlands and the subsequent loss of function. In lagoons and lakes, suspended sediments can reduce water temperature, decrease light availability, impede fish foraging and change the behaviour of macroinvertebrates (Connolly and Pearson, 2007; Wallace et al., 2015). There are also species-specific effects; for example, increased sediment is likely to reduce habitat and interstitial spaces for the freshwater moray of the Wet Tropics (Ebner et al., 2016) and affect the behaviour and impair surface respiration rates in freshwater turtles (Schaffer et al., 2016). Some tropical lowland invertebrate species show resistance to high suspended sediments in riverine wetlands; however, the resistance is short-lived and strongly dependent on the type of sediment (Connolly and Pearson, 2007).

3.2.3 Pesticides

The most commonly detected pesticides in the Great Barrier Reef catchments are (PSII) inhibiting herbicides. These include atrazine, ametryn, hexazinone and diuron, which are mainly used in sugarcane, and tebuthiuron, which is mainly used in rangeland grazing (Lewis et al., 2009). PSII herbicides are reportedly widespread in Great Barrier Reef catchments (Smith et al., 2012; Davis et al., 2013). The frequent detection of diuron, atrazine and hexazinone residues in waterways draining sugarcane in the Tully-Murray River and Barratta Creek indicate detrimental impacts on downstream wetlands (Bainbridge et al., 2009; Davis et al., 2013; O'Brien et al., 2014; Davis et al., 2017).

Herbicides and pesticides are composed of many different compounds with varying levels of toxicity and modes of action (Finlayson and Silburn, 1996). Their toxicity can also change depending on their mixture with other compounds (Magnusson et al., 2010; Smith et al., 2012). Importantly, the effect of pesticides is likely to be highly temporal and associated with the seasonality of the hydrology of wetlands within the Great Barrier Reef catchment (see Chapter 3). Wetland biota may be exposed to acute impacts associated with pulses of high concentrations of pesticides, often greatest in first rainfall or irrigation run-off in the early wet season (Davis et al., 2017). Alternatively, biota can experience chronic impacts associated with low concentrations prevalent throughout the dry season (Davis et al., 2017). Long-term, low concentration exposure effects may not be apparent for several generations (Pesce et al., 2012).

Pesticides have effects not only on the plants of freshwater wetlands, but on the whole food chain. In benthic microalgae, PSII-inhibiting herbicides produce additive toxic effects causing a significant reduction in growth (Magnusson et al*.,* 2008). Herbicides in freshwater wetlands can have impacts on some diatom genera, but not on others (Wood et al., 2014). In amphibians and fish within the Great Barrier Reef, atrazine enhances osmoregulatory disturbance and affects gill histopathology, gonadal morphology, sex hormones, reproductive fitness and population dynamics (Rohr and McCoy, 2009; Kroon et al*.,* 2014a; Kroon et al*.,* 2015b).

3.2.4 Loss of connectivity

For many floodplain wetlands, an overarching factor affecting health is the loss of hydrological connectivity with rivers, floodplains, estuaries and marine ecosystems, which can lead to declines in water quality and biodiversity (Bunn and Arthington, 2002; Arthington, 2012; GBRMPA 2012).

Alterations in the form of artificial barriers in the Great Barrier Reef catchment impede connections for fish and other aquatic fauna (Sheaves et al*.,* 2014; GBRMPA, 2014a). Many fish species use coastal wetlands as nurseries or breeding grounds, and habitat removal or alteration disrupts their life cycles (Meynecke et al*.,* 2008; Sheaves et al*.,* 2014; Harris et al., 2017). In contrast, frequent water flushing of floodplain lagoons in the Tully River has maintained good water quality and relatively high fish biodiversity (Arthington et al., 2014).

In many areas, commercially and recreationally important fisheries species have been excluded through barriers such as tidal exclusion bunds, sand dams, road and rail crossings, other culverts and crossings and macrophyte chokes (Sheaves et al., 2014). The effect of a barrier is determined not just by the characteristics of the structure but also by a range of other factors, including the ecological requirements of the fish at various stages of its life cycle (Harris et al., 2017). Many streams and rivers have weirs at the estuary/freshwater interface preventing connections to upstream habitats (Sheaves et al., 2007; Sheaves et al., 2014). Studies have been conducted to identify barriers to fish passage in many catchments of the Great Barrier Reef, but a lack of consistent baseline data means the effect of these barriers across catchments of the Great Barrier Reef is not easily determined (Stockwell et al., 2008; Lawrence et al., 2009; Lawson et al., 2010; DEEDI, 2011; Moore, 2015; Marsden, 2015; DNPSR, 2016; NQ Dry Tropics, 2016).

Historical records in floodplain wetlands of the Burdekin River show that hydrological changes have altered the diatom community composition, more so than land clearing (Perna et al., 2012a). In contrast, artificial connectivity such as transfer of irrigation water in northern Queensland has provided the means for the spread of exotic species such as tilapia (*Oreochromis mossambicus*; Russell et al., 2003).

The disconnection of wetlands can cause the stagnation of water, the depletion of dissolved oxygen and the accumulation of pollutants that can be rapidly released during large floods causing 'black water' events, which can have significant consequences for downstream ecosystems (Veitch et al., 2008). Disconnected systems can still retain biodiversity values. For example, isolated and highly modified wetlands in the Burdekin delta support some native fish species (Davis and Moore, 2015), and remnant riparian wetlands are important for sustaining the biodiversity of birds in the Wet Tropics (Keir et al., 2015).

3.2.5 Climate change

Climate change will increase temperatures and modify wetland hydrology as a result of variations in the timing, frequency and intensity of rainfall (CSIRO and BoM, 2015). Climate change will also result in sea level rise, increasing saline intrusions into coastal freshwater wetlands (Williams et al., 2012).

Temperature and oxygen concentrations are highly linked as high temperature decreases the capacity of oxygen to dissolve in water. The impact of changes in temperature and dissolved oxygen will depend on the characteristics of each wetland, as well as season. During warm seasons, freshwater wetlands can experience low dissolved oxygen concentrations which can contribute to asphyxiation of aquatic animals that are not able to surface respire or escape (Burrows et al., 2006; Waltham and Davis, 2016). For many aquatic species, including fish, temperature directly controls metabolic rate, but it can also influence growth, resource allocation for reproduction and, ultimately, population size (Jobling, 1995). Thermal risks may contribute to both acute and chronic exposure effects influencing habitat suitability for aquatic biota (Waltham et al., 2013; Wallace et al., 2015). Changes in temperature and oxygen during dry and wet periods are major contributors to the biodiversity, range shifts, species distribution and extinction of freshwater wetlands in the region (Burrows and Butler, 2007, 2012; James et al., 2013; James et al., 2017).

Variations in the timing, frequency and intensity of rainfall as a result of climate change will have profound effects on the hydrology of wetlands, and thus on their ecological functions (Mitsch et al., 2010). The frequency and intensity of dry and wet periods will modify wetland communities within the Great Barrier Reef catchment (Brock et al., 2003; Donaldson et al., 2013).

Finally, salinisation of freshwater wetlands near the coast is likely to occur as a result of sea level rise and may drastically change wetlands as mangroves invade saltmarsh (Saintilan and Rogers, 2013) and possibly *Melaleuc*a forests. The alterations in vegetation composition due to increased salinity will result in changes in many ecological functions, such as carbon sequestration (Kelleway et al., 2016; Tran and Dargusch, 2016).

3.2.6 Weed infestation

Weed infestation is one of the most common and evident impacts on freshwater wetlands in the Great Barrier Reef catchment. Floating aquatic weeds typically flourish in tropical and subtropical locations (Howard and Harley, 1997), and many coastal wetlands in the Great Barrier Reef catchment are infested with them (Perna et al*.,* 2012b). Floating plants such as water hyacinth and emergent grasses such as para grass cause severe degradation in northern Australian wetlands (Pusey and Arthington, 2003). Two of the most prevalent and highest threats to wetland function in the Wet Tropics include para grass (*Brachiaria mutica*) and Hymenachne (*Hymenachne amplexicaulis*) (Werren, 2001). The increase in weeds has cascading effects in primary and secondary production and in the overall function of freshwater wetlands (Adame et al., 2017).

Weed infestation causes the degradation of water quality in tropical regions by reducing oxygen levels in the water column through the decomposition of organic matter (Perna and Burrows, 2005; Adame et al., 2017). The decrease in dissolved oxygen can have negative effects on fish and invertebrates (Pusey and Arthington, 2003; Perna et al., 2012b). For example, water hyacinth and *Salvinia* can develop dense mats that significantly reduce light penetration into the water, thereby inhibiting photosynthesis in submerged plants and reducing oxygen (Perna and Burrows, 2005). Dissolved oxygen concentrations strongly determine secondary productivity of wetlands in the Great Barrier Reef region (Burrows et al., 2006; Burrows and Butler, 2007; Butler and Burrows, 2007). Hypoxia can affect many fish species, but their tolerance varies among species and life stages (Butler and Burrows, 2007). Weed infestation affects the biodiversity of freshwater wetlands through their effects on dissolved oxygen and temperature, which in turn affect fish populations.

While weeds cause the degradation of water quality, poor water quality favours the establishment of weeds (Brodie and Mitchell, 2005; Davis et al., 2017). Weed growth is stimulated by inflow of nutrient rich water, such as run-off after periods of rainfall, particularly in static waters (Howard and Harley, 1997). The growth of dense patches of weeds facilitates the retention of suspended particles, enabling further plant colonisation (Arthington et al., 1997). Furthermore, the alteration of hydrology in areas with crop irrigation affects the seasonal growth patterns of native plant species, further facilitating the proliferation of weeds (Connolly et al., 2015). Thus, the degradation of wetlands favours the establishment of weeds, which results in further deterioration of the water quality and biodiversity of the wetland.

3.2.7 Vertebrate pests

There are a number of aquatic and terrestrial pests of national significance in wetlands of the Great Barrier Reef region, including pigs, tilapia, gambusia guppies, swordtails and platys (Harrison and Congdon, 2002). However, there is a lack of data on the abundance and distribution of these species, and their effects on wetlands in the Great Barrier Reef catchments are largely unassessed (Harrison and Congdon, 2002).

Feral pigs (*Sus scrofa*) can be abundant in and around wetlands as they tend to concentrate near water (Waltham and Schaffer, 2016). They can cause major damage to freshwater wetlands; in Lakefield National Park on Cape York, pigs have changed the aquatic macrophyte communities, increased nutrients and water turbidity (Doupe et al., 2010). These changes have caused prolonged anoxia and pH changes in the water column, all of which have affected the habitat of freshwater turtles (Doupe et al., 2009). The introduction of feral pigs into tropical wetlands of Australia might have cascading effects within the trophic chain.

Infestation of tilapia is also prevalent in many wetlands of the Great Barrier Reef catchments, including the Mulgrave, Barron, Burnett and Burdekin catchments (Russell et al., 2012). Tilapia can displace native species, modify their habitat and introduce diseases in natural fish populations (Russell et al., 2012).

4. Condition of estuarine ecosystems of the Great Barrier Reef region

4.1 Changes in estuarine ecosystem extent

In 2013[26], the Great Barrier Reef region contained 4333 km^2 of estuarine ecosystems (Wetland*Info*, 2016). This represents 92.5% of the pre-European extent (Wetland*Info*, 2016). Since 2001, the loss of estuarine wetlands has been low (<0.1%) in all regions except in the Fitzroy NRM region, where wetland loss increased to 260 ha (0.3%).

Changes in hydrology and rainfall patterns (Eslami-Andargoli et al., 2010), and acute pressures (such as climatic events) are compounded by the chronic stress of poor water quality and can drive localised dieback of mangrove forests in the Great Barrier Reef region (Lovelock et al., 2009). A significant proportion of the Queensland coast has been developed since European settlement, impacting on estuarine wetlands, with the most significant contributions being drainage and land reclamation and the ensuing loss of habitat (Wegscheidl et al., 2015). Although attributing mortality in mangrove stands to a single factor is overly simplistic, there are effects that can be predominantly associated with one or more impacts. These impacts will be assessed in the following sections.

4.2 Water quality and other pressures affecting the condition of estuarine ecosystems

4.2.1 Nutrients

Estuarine wetlands are usually limited by nitrogen and/or phosphorus, and nutrient enrichment can enhance plant growth (Reef et al., 2010; Reef et al., 2016). However, in mangroves, the benefits of increased growth in response to increased nutrient levels can be offset by high mortality rates and slow recovery rates during periods of drought and after tropical storms (Lovelock et al., 2009; Feller et al., 2015). This is because nutrients, especially nitrogen, stimulate the growth of shoots relative to

[26] Only information to 2013 was available at the time of writing.

roots, which causes physical instability of the mangrove trees (Lovelock et al., 2009). The decrease in root biomass can also enhance subsidence of soils, decreasing the capacity of the forest to keep pace with sea level rise (McKee et al., 2007).

Given the response of mangroves to nitrogen, high nutrient loads delivered in floodwaters to the Great Barrier Reef could be a contributor to the localised dieback of mangrove forests when followed by periods of low rainfall and, thus, high salinity (Eslami-Andargoli et al., 2010) or after tropical storms, such as after TC Yasi in 2011 (Asbridge et al., 2015; Feller et al., 2015). During a period of very low rainfall, mangroves in Port Douglas and Hinchinbrook Channel had significantly greater canopy loss in trees fertilised with nitrogen at the most saline site (Lovelock et al., 2009).

Nutrient enrichment also has the potential to favour algal blooms in estuarine wetlands, which can smother pneumatophores and seedlings and result in the shift of primary production from a plant to an algae-dominated system, which is sometimes an irreversible change (Verhoeven et al., 2006). Finally, nutrient enrichment can change the microbial community in estuarine sediments (Adame et al., 2012a).

Estuarine wetlands have a relatively high capacity for nutrient retention in low and medium flood events and can play a role in protecting the marine environment from land-derived nutrient pollution (Valiela and Cole, 2002; Alongi and McKinnon, 2005; Adame et al., 2010a). However, in the long term, nutrient enrichment may have negative consequences affecting vegetation structure and composition and reducing their capacity for nutrient retention (Verhoeven et al., 2006; Reef et al., 2010).

4.2.2 Sedimentation

Estuarine ecosystems are important depositional zones along the Queensland coast (Adame et al., 2010a; Adame et al., 2012b) typically dominated by fine soft sediments (Furukawa and Wolanski, 1996). Urbanisation and agriculture have increased sediment loads delivered to estuarine ecosystems in the Great Barrier Reef region, adversely affecting their biodiversity, ecosystem structure and function (Thrush et al., 2004).

Increased sedimentation has led to the expansion of mangroves in some areas of the Great Barrier Reef. For example, mangrove expansion in Trinity Bay during the last century has been associated with an increase in the deposition of unconsolidated sediments (Wolanski and Duke, 2002). Excessive sedimentation may increase mangrove areas in some locations but can also cause immediate death by burying plants and seedlings and covering pneumatophores (Ellison, 1998; Saintilan and Williams, 1999; Wolanski and Duke, 2002; Lovelock and Ellison, 2007).

Low sediment supply, as can occur during long periods of drought or in rivers with large dams, can also have detrimental impacts on intertidal wetlands (mangroves and saltmarsh). The ability of coastal wetlands to keep pace with sea level rise is partly due to their capacity to accumulate sediment (Krauss et al., 2013; Woodroffe et al., 2016). For example, in south-east Queensland, wetlands with fine sediment require high sediment deposition to accrete (Lovelock et al., 2013). Mangroves may overcome sporadic episodes of high sediment supply as occurs during high floods (Davies et al., 2016) and can keep pace with sea level rise if sediment supply is neither too high nor too low.

4.2.3 Pesticides

Estuarine wetlands are susceptible to the delivery of pesticides as they receive pulses of water directly from agricultural lands during periods of high rainfall (Davies et al., 2016). 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 in ambient conditions. Long-term studies of chronic exposure of mangroves to herbicides and other co-occurring factors at biologically relevant concentrations are a key knowledge gap.

4.2.4 Sea level rise

An important factor impacting estuarine wetlands in the future is sea level rise (Lovelock and Ellison, 2007). The capacity of estuarine wetlands to keep pace with sea level rise depends on productivity, sediment supply, grain size of the sediment, tidal range and rainfall (Lovelock et al., 2011a; Lovelock et al., 2013; Lovelock et al., 2015a). In the Great Barrier Reef, it can be expected that intertidal wetlands in the mouth of rivers with high availability of fine sediment will rely on episodes of high rainfall to accumulate sediment, while wetlands on islands or in sandy sediment will depend on root growth (Lovelock et al., 2015b). Vegetation composition will also affect the capacity of estuarine wetlands to accumulate sediment. For example, in south-east Queensland, saltmarshes dominated by *Juncus kruassii* had higher rates of soil accumulation compared to marshes dominated by *Sarcocornia quinqueflora* (Lovelock et al., 2013). In the Hinchinbrook Channel, root production of mangroves is very high, especially in fringing forests dominated by *Rhizophor*a (Robertson and Alongi, 2016), which might help these communities keep pace with sea level rise. A model for the Indo-Pacific predicts that mangroves with low tidal range and low sediment supply are the most susceptible to be submerged by 2100 (Lovelock et al., 2015a). According to the model, mangroves from the Great Barrier Reef are at low risk of being submerged by 2100, although mangroves north of Cooktown have a relatively higher risk (Lovelock et al., 2015a).

If mangroves do not keep pace with sea level rise, the fringe forest—usually the tallest portion of the forest—will be lost (Gilman et al., 2008). Some landward mangroves may expand into the saltmarsh or into adjacent freshwater wetlands where these are co-located (Eslami-Andargoli et al., 2010; Saintilan and Williams, 1999). In the past decade, mangroves have been encroaching on saltmarshes throughout the Queensland and New South Wales coast, and in many locations saltmarshes are already at risk of being lost (Saintilan and Rogers, 2013). Saltmarshes in the Great Barrier Reef area are highly vulnerable to climate change due to their species-poor assemblage and their relatively low capacity to migrate landward (Bridgewater and Cresswell, 1999; Saintilan and Rogers, 2013). Changes in the vegetation as sea level rises are likely to have consequences for ecosystem function, such as in the diversity and abundance of macroinvertebrates as food for shorebirds (Sheaves et al., 2016a). The incursion of mangroves into saltmarshes might increase the carbon stored in intertidal systems (Kelleway et al., 2016). However, in the Great Barrier Reef region, mangroves might be limited in their capacity to move landward as they are frequently bordered not by saltmarsh, but often by agricultural land, roads or steep hillslopes.

4.2.5 Climate change

Expansion and contraction of mangroves and saltmarshes are sensitive to climatic changes (Jupiter and Marion, 2008). Changes in the frequency and intensity of rainfall will have major effects on estuarine wetland through modulation of porewater salinity, nutrient availability and sediment supply (Gilman et al., 2008).

The higher rainfall associated with tropical areas in Australia shapes the structure and function of mangrove communities (Ball, 1998), while prolonged inundation may ‘suffocate’ mangroves (McKee, 1996). A significant positive relationship has been demonstrated between median rainfall and landward expansion of mangroves into saltmarshes in south-east Queensland (Eslami-Andargoli et al., 2010). Increased rainfall can also increase nutrient uptake in mangroves and cyanobacterial salt flats (Adame et al., 2010b; Adame et al., 2012b). Conversely, a long-term decline in rainfall in the Fitzroy catchment has been associated with a shift in vegetation from mangrove-dominated communities to saltmarsh (Duke et al., 2003).

Tropical storms and cyclones can have acute destructive impacts on estuarine wetlands, such as defoliation, abrasion, stem breakage, uprooting and smothering by sediment (Woodroffe and Grime, 1999; Paling et al., 2008). However, tropical storms can also increase post-disturbance growth rates through alleviation of hypersalinity and increases in nutrients (Lovelock et al., 2011b). In 2011, TC Yasi damaged 17% of the mangroves in the Hinchinbrook Channel, especially those in the seaward fringe (Asbridge et al., 2015). Significant recovery of the damaged forest was observed after one year of the cyclone (Asbridge et al., 2015).

The predicted increases in the frequency and intensity of extreme weather events may alter the community structure and productivity of mangroves in the Great Barrier Reef region (Lovelock et al., 2016). It is possible that frequent tropical storms will favour species more tolerant of disturbances through rapid regeneration (Baldwin et al., 2001). Increases in atmospheric CO_2 will also have impacts on estuarine wetland vegetation. For example, rising CO_2 will increase production with limits set by changes in salinity and nutrients as a result of rising seas (Reef et al., 2015). In areas where nutrients and salinity are high, mangroves of the *Avicennia* genus may become dominant (Reef et al., 2016). It is predicted that an overall decrease in precipitation and an increase in sea level of 18−20 cm in Australia has already impacted mangroves across Australia (Alongi, 2015). The increase in sedimentation rates, increases in CO_2 and the plasticity of mangroves to tolerate variability in salinity and temperature might result in mangrove expansion in the coast of the Great Barrier Reef (Alongi, 2015; Lovelock et al., 2016). However, it is likely that the mangroves will go through a shift in species composition and structure, and a loss of saltmarsh habitat as mangroves potentially migrate landward may occur. Encroachment of intertidal wetlands may follow salinisation, with a dieback of freshwater marshes and *Melaleuca* forests. Further acceleration of sea level rise, nutrient enrichment and sedimentation may result in the loss of intertidal wetlands of the region (Duke et al., 2007; Lovelock et al., 2015a).

4.2.6 Loss of connectivity

One of the greatest threats to estuarine wetlands is loss of connectivity, mainly driven by the loss of hydrological connections between rivers, freshwater wetlands and marine ecosystems. For example, in the Tully River, hydrological connections between fresh and estuarine wetlands have been reduced by riverbank elevation and levee banks (Karim et al., 2012). Changes in rainfall are also highly likely to affect hydrological connectivity and, thus, the productivity and diversity of estuarine wetlands. In Queensland, the significance of connectivity of estuarine wetlands has been linked to inshore fisheries production (Meynecke et al., 2008), the composition of estuarine fish communities (Sheaves and Johnston, 2008) and the capacity of wetlands to keep pace with sea level rise (Lovelock et al., 2011a).

The full impact of lost connections is difficult to interpret. In mangroves of the Hinchinbrook region, few species of fish regularly used mangrove forests (Sheaves et al., 2016b). In estuaries adjacent to the Great Barrier Reef, mangrove extent along with catchment hydrology and substrate and estuary mouth configuration had significant impacts on fish assemblages (Ley, 2005). The value of intertidal wetlands in the Great Barrier Reef catchment cannot be directly translated as habitat use (Sheaves et al., 2015); rather, it is the range of ecological processes associated with them, such as the export of mangrove-derived nutrients and carbon, as well as the provision of temporal refugia that defines their true value (Sheaves et al., 2015; Wegscheidl et al., 2015).

Estuarine wetlands are also habitat for native mammals, migratory birds, fish, crustaceans and other invertebrates (Cannicci et al., 2008; Nagelkerken et al., 2008; Saintilan and Rogers, 2013; Sheaves et al., 2015). The loss of connections to terrestrial and marine habitats will likely affect not only wetland, but also terrestrial and marine biodiversity within the Great Barrier Reef region.

4.2.7 Other factors

Direct physical disturbances on intertidal wetlands is a problem in many areas of the Great Barrier Reef region. For example, vehicle transit in saltmarshes has a large impact in some locations (Trave and Sheaves, 2014), such as in the Halifax wetlands in the Herbert River catchment.

5. Seagrass ecosystems

5.1 Condition and trends of Great Barrier Reef seagrass meadows

Seagrass meadows are a critical component of the Great Barrier Reef inshore ecosystems, supporting charismatic megafauna such as dugong and turtles and providing ecosystem services that make them a high conservation priority (Costanza et al., 1997; Marsh et al., 2011; Waycott et al., 2011; Cullen-Unsworth and Unsworth, 2013; Costanza et al., 2014; Coles et al., 2015). Much of the connectivity in reef ecosystems depends on intact and healthy seagrass meadows (Waycott et al., 2011), as they provide habitat for maintenance and regeneration of reef fish populations, in addition to locally mitigating the effects of ocean acidification (Fourqurean et al., 2012; Unsworth et al., 2012) and improving water quality by controlling pathogenic bacteria to the benefit of marine invertebrates such as coral (Lamb et al., 2017).

Although the greatest area of seagrass in the Great Barrier Reef World Heritage Area (modelled area is 32,335 km^2) is located in waters deeper than 15 m in the Great Barrier Reef lagoon (Coles et al., 2009; Carter et al., 2016; McKenzie et al., 2010), these deep meadows are relatively sparse and the source of minor ecosystem services (Derbyshire et al., 1995; McKenzie et al., 2010; Macreadie et al., 2014). In shallower areas <15 m, seagrasses proliferate under optimal conditions and approximately 3464 km^2 of meadows have been mapped since 1985 (McKenzie et al., 2014; Saunders et al., 2015; Carter et al., 2016; McKenzie et al., 2016a; Waterhouse et al., 2016). It is these shallow, inshore meadows that are of greatest conservation concern, particularly south of Cooktown. They are the most productive seagrass units but are also adjacent to the developed (agricultural/urban) coast, which makes them vulnerable to land-based threats, including agricultural run-off and coastal development (Grech et al., 2011).

Monitoring since 1999 indicates that inshore seagrass meadows along the developed coast of the Great Barrier Reef have fluctuated dramatically in abundance (above-ground per cent cover) as a consequence of various extreme events (Figure 7). Widespread impacts to seagrass along the Great Barrier Reef developed coast from multiple years of above average rainfall and extreme weather events (in 2009-2011) led to a decline in abundance in all regions (Figure 7). Despite reduced pressures, inshore seagrass meadows along most of the Great Barrier Reef have only recovered slowly. There has been some recovery of abundance in Cape York, Mackay Whitsunday and Burnett Mary regions, while abundance in the Burdekin region has shown substantial recovery (Davies et al., 2016; McKenzie et al., 2016b). A number of meadows in the Wet Tropics have failed to recover their abundances in 2015 (McKenzie et al., 2016b), while others including Cairns Harbour have shown signs of recovery (York et al., 2016). In the Fitzroy region ongoing disturbances, including TC Marcia, have delayed recovery. By contrast, the meadows to the immediate north of the Great Barrier Reef in the Torres Strait have remained relatively stable over similar time frames (Carter et al., 2014; Carter et al., 2015; Sozou et al., 2016).

These impacts also left a legacy of reduced resilience with weak resistance to further disturbances and a low capacity to recover from further losses (McKenna et al., 2015; McKenzie et al., 2016b). Indicators of recovery are improved trajectories in seagrass abundance and greater meadow extent, but resistance remains weak as abundances have not fully recovered and the contribution of the persistent species to community composition remains low. Reduced reproductive effort and low seed densities at most locations have been implicated in the slow recovery (Davies et al., 2016; Davies and Rasheed, 2016; McKenna et al., 2016a; McKenna et al., 2016b; McKenzie et al., 2016b;

Reason et al., 2016; York et al., 2016). There are indications also that water quality (e.g. light levels) has deteriorated within some seagrass meadows and that this, coupled with higher water temperatures, may have restricted seagrass growth in some meadows in all Great Barrier Reef regions (Collier et al., 2011; Negri et al., 2015; Collier et al., 2016a; McKenzie et al., 2016b; Adams et al., 2017). The natural resource management regions of greatest concern are the Fitzroy and Wet Tropics (especially from south of Cairns to Hinchinbrook Island), where the indicators of seagrass resilience (abundance, reproductive effort, seed bank, species composition) remain very low, rendering the meadows highly vulnerable to further stressors (York et al., 2015a; McKenzie et al., 2016b).

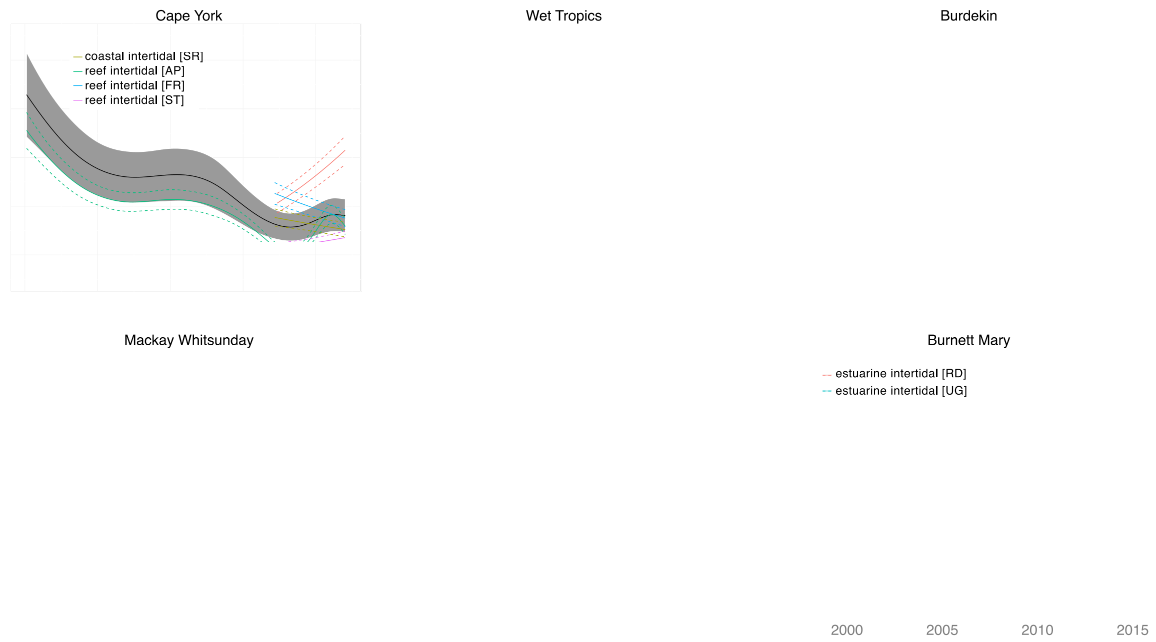

Figure 7: Regional inshore seagrass abundance trends (in per cent cover), represented by a Generalised Additive Model plot, and trends for each location and habitat type from 1998 to 2015. (Source: McKenzie et al., 2016b). Shading or dashes show 95% confidence intervals around each trend line.

5.2 Water quality and seagrass condition

When adequate light is available to maintain growth, the overarching factors influencing seagrass distribution are physical (waves, currents, tides, bioturbation and grazing), geological (sediment grain size and organic matter) and geochemical (sediment mineralogy, bioavailability of nutrients and phytotoxins) (Koch, 2001; Hebert et al., 2007; Grech and Coles, 2010). The interaction of these factors with local water quality results in complex and dynamic seagrass habitats (Waycott et al., 2005; Collier and Waycott, 2009). Colonisation depth and abundance are generally controlled by water quality (Abal and Dennison, 1996; Collier et al., 2012a; Waycott and McKenzie, 2010; McKenzie et al., 2012a). Frequent exposure to turbid water, with low benthic light, caused widespread declines in seagrass abundance and condition from 2009 to 2011 (McKenzie et al., 2012a; Rasheed et al., 2014; Petus et al., 2014b; Petus et al., 2016). The major water quality pollutants that undermine seagrass resilience are fine sediment, elevated nutrients and herbicides (Brodie et al., 2017a).

5.2.1 Light and turbidity

Seagrass photosynthetic carbon fixation increases with light availability at sub-saturating light levels, and carbon fixation controls growth and biomass production. When light availability is reduced in the short term, photosynthetic rates decline and result in metabolic changes that include drawdown of storage reserves and slower growth rates. Both responses enable seagrasses to survive below their minimum light requirements for short periods (Ralph et al., 2007). These early responses to light limitation have been detected in one species (*Zostera muelleri*) as changes in the expression of reference genes (Schliep et al., 2015); however, further research is required to establish their potential as a sublethal indicator of light limitation. If light limitation is sustained (for weeks or months) there may be morphological changes such as increased 'senescence' (shedding of leaves on shoots, thus reducing abundance), eventual plant death and decreased spatial distribution, particularly contraction of the depth range (meadow-scale changes) (Chartrand et al., 2016; Collier et al., 2016a; Collier et al., 2012b). Most *Halophila* spp. are adapted to low light and occupy habitats that have naturally low light levels (e.g. deepwater). However, they are also very sensitive to light limitation (when light falls below thresholds), quickly shifting from metabolic to meadow-scale impacts after just two to four weeks below light thresholds (2–6 mol/m^2/d; Collier et al., 2016b). By contrast, structurally larger strap-leaved species, which have higher light requirements and are impacted at higher light thresholds (5–6 mol/m^2/d; Collier et al., 2016b), are able to tolerate months of light limitation by changing metabolic processes to draw on reserves (Collier et al., 2012b; Collier et al., 2016b). Light thresholds are used to assess the risk of short-term light deprivation (McKenzie et al., 2016b). The minimum light requirements (defined as the light required to maintain productive meadows over one year) are not known for most Great Barrier Reef seagrasses.

Water quality affects the light-attenuating properties of the water and therefore benthic light, which is required for seagrass photosynthesis. Benthic light was below the light threshold for inshore species that indicates light limitation (6 mol/m^2/d; Collier et al., 2016b) for an average of 17.6% of days annually (McKenzie et al., 2016b). In 2014-2015, light thresholds were not met most frequently in the Burdekin NRM region (36% of days) followed by Mackay Whitsunday (25.7%), Burnett Mary (18.5%), Fitzroy (17.2%), Wet Tropics (13.8%), and the least often in Cape York (11.3%) (McKenzie et al., 2016b). In 2014-2015, daily light was slightly lower than the long-term average revealed by monitoring at many Great Barrier Reef locations, particularly those where plankton and coloured dissolved organic matter in secondary water types (as characterised by Petus et al., 2016) reduced not only the quantity but also the spectral quality of light (McKenzie et al., 2016b). Of greatest concern, however, is the long-term impact of light limitation on seagrass shoot densities and growth rates with increasing temperatures, which can result in critical metabolic imbalances (reduction in photosynthesis and increase in leaf respiration), reducing growth rates and abundance (e.g. biomass) (Collier et al., 2016a; Collier et al., 2011).

5.2.2 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 form most readily absorbed by both roots and leaves of seagrasses (Romero et al., 2006). Sources of dissolved inorganic nutrients for inshore seagrasses include land run-off and the biogeochemical transformation processes of organic nutrients in the sediment and water column (Furnas et al., 2011). While additional nutrients can enhance seagrass growth (Udy and Dennison, 1997; Udy et al., 1999), they also favour the growth of plankton, macroalgae and epiphytic algae, which attenuates the light available to seagrass leaves. Further nutrient enrichment can result in an ecosystem state shift away from seagrass-dominated towards macro- or microalgae-dominated benthos (Wazniak et al., 2007).

Levels of leaf nitrogen and phosphorus have declined in inshore seagrasses since 2010, although carbon:nitrogen ratios of <20 indicate that nitrogen loads still exceed biological demand at coastal, reef and estuarine intertidal sites (McKenzie et al., 2016b). There is no evidence that these excessive loads are directly detrimental to seagrass condition (Waycott et al., 2005), although nutrient loading has been linked with the growth of epiphytes (fouling algae) on the leaves of seagrasses. The occurrence of epiphytes has been highly variable since records began in 2005; however, at subtidal sites in the Wet Tropics and Burdekin regions there have been increased epiphyte burdens since

2014 (McKenzie et al., 2016b). Since 2011, seagrass meadows in the Wet Tropics, Burdekin and Mackay Whitsunday regions have also endured high frequency of exposure to green, phytoplankton-rich water (secondary water type; Lønborg et al., 2016; McKenzie et al., 2016b), indicating high bioavailability of nutrients. Epiphyte loads and exposure to green secondary-type water interact to exacerbate light stress in seagrass meadows.

5.2.3 Pesticides

Inshore seagrass meadows of the Great Barrier Reef are exposed to herbicides (Haynes et al., 2000; Lewis et al., 2009), in most cases to mixtures of compounds (Lewis et al., 2012). Although the exposure is year-round, it is greatest during the wet season (Gallen et al., 2016). The PSII-inhibiting herbicide diuron, which is commonly found in Great Barrier Reef inshore waters, adversely affects seagrass photosynthetic efficiency at ambient concentrations in the Great Barrier Reef lagoon (<0.01−1.7 µg/L; Lewis et al., 2012) and at concentrations well below the Great Barrier Reef guideline (0.9 µg/L; GBRMPA, 2010). Specifically, photosynthetic efficiency of seagrass is inhibited by 10% at 0.47–0.49 µg/L of diuron and by 50% at 2.41–2.47 µg/L (Flores et al., 2013). Furthermore, herbicides occur in mixtures (Lewis et al., 2012; Gallen et al., 2016) and these have an additive effect on seagrass photosynthetic efficiency (Wilkinson et al., 2015a). Seagrasses are also sensitive to chronic exposure of low herbicide concentrations, causing measurable impacts to photosynthetic carbon assimilation (≥0.6 µg l^{-1} diuron equivalents) and energetic status (≥1.7 µg/L diuron equivalents) that may leave the plants vulnerable to other simultaneous stressors such as reduced light (Negri et al., 2015). High concentrations of diuron and atrazine may have contributed to localised losses of seagrass meadows in the past, such as in Sarina Inlet in 2011 (Kennedy et al., 2012; McKenzie et al., 2012b). However, the overall contribution to widespread seagrass losses from 2009 to 2011 due to the cumulative impacts of herbicides and low benthic light from multiple years of above average rainfall and river discharge is unknown.

5.3 System-wide flow-on effects of seagrass decline

The responses of seagrasses in the Great Barrier Reef region to chronic changes in water quality still remain 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 (e.g. Collier et al., 2012a; Collier et al., 2014; Wilkinson et al., 2015b; York et al., 2015b; Chartrand et al., 2016). It is likely that chronic declines in water quality can shift seagrass meadow distributions and result in changes to community composition (Kilminster et al., 2015). 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 (Unsworth et al., 2015).

The loss of seagrass from reduced water quality and physical disturbance as a result of floods and tropical cyclones is known to have significant flow-on effects to dugong and green turtle populations, which are highly dependent on seagrass meadows for their primary food supply (Preen and Marsh, 1995; Marsh et al., 2011; Meager and Limpus, 2012). Malnutrition makes these herbivores prone to disease, and lack of food may force them to move long distances to find alternative sources. As a consequence of the widespread loss of seagrass along the developed coast of Queensland in early 2011, stranding rates of sea turtles and dugong increased dramatically during that year throughout the Great Barrier Reef and produced the highest mortality rates since records began in 1997 (Figure 8). As seagrass abundance improved in many regions (see above), in 2015, dugong mortalities decreased to the lowest levels since records commenced. In contrast, turtle mortalities have declined since 2011 but remain above the long-term median. The flow-on effects from seagrass loss to other associated fauna and/or multi-species fisheries are less obvious than in the case of the major herbivores; they may manifest as changes in community composition rather than losses of biomass. For example, decline in one species of penaeid prawn caused by seagrass loss was balanced by increase in the biomass of other species of prawns (Connolly et al., 1999), although it is questionable whether they present a similar economic value.

The consequences of seagrass loss are not limited only to associated fauna. As seagrasses influence their physical, chemical and biological surroundings (i.e. they are environmental engineers) their decline can have broader consequences related to coastal processes including carbon capture and storage, nutrient dynamics, sediment stabilisation and habitat connectivity (Waycott et al., 2007; Adams et al., 2016; Maxwell et al., 2016).

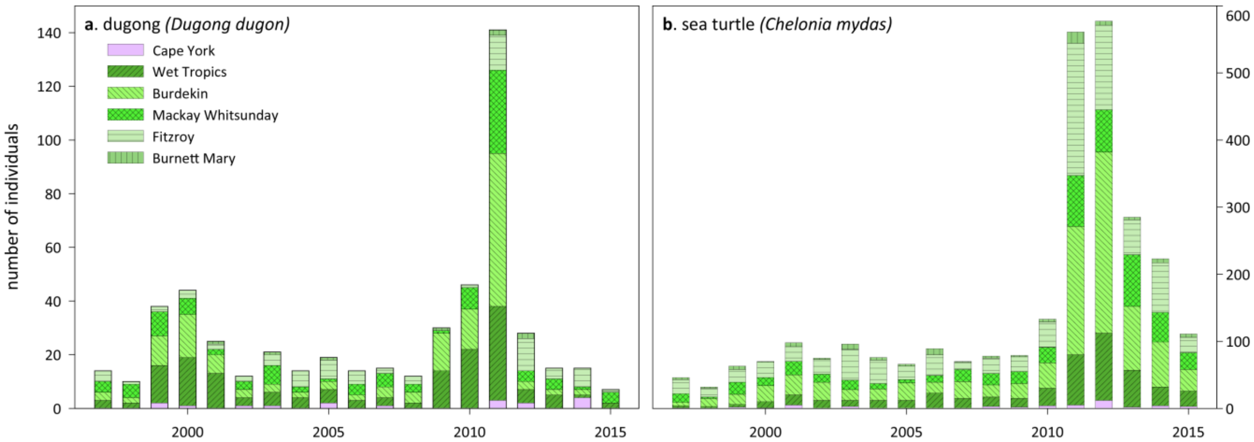

Figure 8: Dugong and sea turtle mortalities (as number of stranded animals per year) along the Great Barrier coast attributed to natural or unidentified causes in each natural resource management region from 1997 to 2015. (Source: data courtesy StrandNet, accessed 4 October 2016). Mortalities do not include boat strikes, drownings or hunting.

6. Coral reef ecosystems

6.1 Condition and trends of Great Barrier Reef coral reefs

It is well recognised that coral reefs go naturally through cycles of disturbance and recovery, and the observed long-term trends generally reflect the regional histories of disturbance. An updated analysis of the Australian Institute of Marine Science Long-term Monitoring Program data to 2015[27] showed significant recovery from 2012 to 2015 on reefs south of about Cairns after the reported long-term decline in hard coral cover (De'ath et al., 2012). However, reefs further north showed a new decline in coral cover over the same period because of an intense tropical cyclone (a second tropical cyclone occurred after the analysed survey period) and renewed activity of crown-of-thorns starfish in the region.

The most recently published assessment of the condition of Great Barrier Reef inshore reefs as part of the Marine Monitoring Program indicates improvements in the coral index in recent years (Figure 9) coinciding with low levels of run-off entering the Great Barrier Reef lagoon due to the recent run of dry years (Thompson et al., 2016). These regularly monitored inshore reefs range from Snapper Island (close to the Daintree River, Wet Tropics region) to Keppel Bay (adjacent to the Fitzroy River, Fitzroy region), but do not include sites in the Cape York or Burnett Mary regions. The Marine Monitoring Program assessment concluded that adverse responses of coral communities occur when they are exposed to extremes in environmental conditions beyond those to which they are either adapted or acclimated (see also Thompson et al., 2014).

[27] www.aims.gov.au/docs/media/latest-news/-/asset_publisher/EnA5gMcJvXjd/content/05-april-condition-of-great-barrier-reef-corals-before-the-mass-bleaching-event-in-2016

Coral reefs in the Torres Strait region are not covered by regular monitoring activities, and trend information is not available. Comprehensive surveys of coral and reef fish biodiversity were completed at reefs surrounding nine Torres Strait islands in 2013 and 2014 (Bainbridge et al., 2015; Sweatman et al., 2015). The surveyed coral reefs were in good condition with high coral biodiversity and moderate to high coral cover compared to Great Barrier Reef reefs. Some outbreaks of crown-of-thorns starfish were observed as well as some bleaching-related mortality.

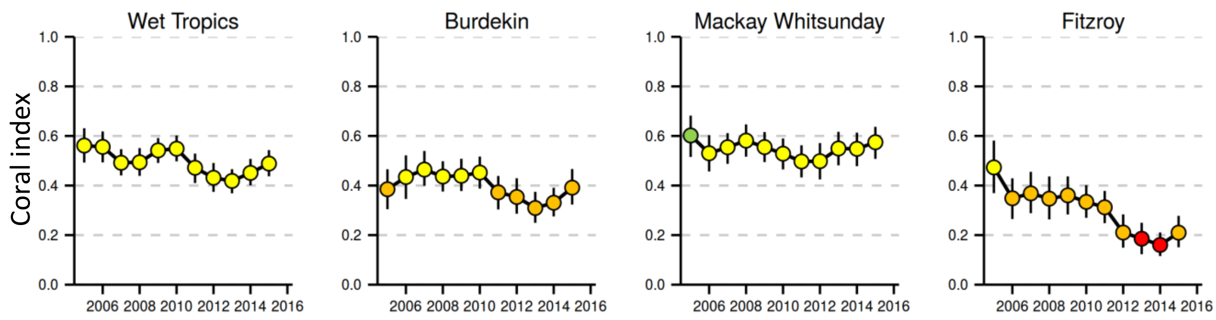

Figure 9: Regional coral index as reported by the Marine Monitoring Program. (Source: Thompson et al., 2016). The regional coral index aggregates scores for coral cover, proportional cover of macroalgae, density of juvenile corals, the rate of coral cover increase and change in coral community composition. Colours for the coral index classify condition: red = very poor, orange = poor, yellow = moderate, light green = good, dark green = very good**.**

At the southern end of the Great Barrier Reef, adjacent to the Burnett Mary region, inshore fringing reef coral communities exist at the southern limit for consolidated reef formation along the mainland of eastern Australia (Coppo et al., 2014). Some of these reefs are surveyed on an irregular basis. After reports of relatively high coral cover on some reefs, flooding of the Mary River in 2011 resulted in significant coral mortality and reduced coral cover at these reefs (Butler et al., 2013). This followed exposure of the reefs to three months of waters with low salinity, high turbidity and potentially elevated nutrients. Further mortality was observed after severe flooding of the Mary and Burnett rivers in 2013. It is expected that these reefs will recover over the next few years, if not further disturbed, based on observations of recovery from previous flood-related mortality in 1974, 1992 and 1999 (Coppo et al., 2014).

It should be emphasised that the reef condition assessments above have been based on data from before the mass thermal bleaching event that affected the Great Barrier Reef in 2016. The most severe bleaching occurred in the Far Northern Great Barrier Reef, from Port Douglas to the Torres Strait Islands, and has resulted in significant coral mortality across many species (GBRMPA, 2016a; Hughes et al., 2017). Recent coral cover data from the long-term monitoring sites of the Long-term Monitoring Program and Marine Monitoring Program are being analysed and updated; trend information was not published at the time of writing. The inshore reefs monitored as part of the Marine Monitoring Program appear to have escaped major coral mortality during the 2016 summer (Thompson et al., in review).

In an environment of frequent disturbances, the persistence of coral communities depends on resilience, which is the product of resistance to pressures and ability to recover during periods of low disturbance (Anthony et al., 2015). For example, the stress associated with acute bleaching may impair calcification rates (Cantin and Lough, 2014) and coral reproduction for several years (Carilli et al., 2009). This is discussed in more detail in the last section of this chapter. While this Scientific Consensus Statement focuses on the effects of water quality, it is becoming increasingly clear that the cumulative impacts of multiple pressures are shaping coral reef communities, which are often a combination of local or regional water quality pressures and global pressures, such as increasing temperature and ocean acidification (Uthicke et al., 2016).

6.2 Water quality and reef resilience

Poor water quality as a result of the loads of sediments, nutrients and pesticides delivered to the inshore reefs in catchment run-off has the potential to both increase the susceptibility of corals to disturbances and suppress their subsequent recovery. The 2013 Scientific Consensus Statement concluded that the location of reefs is one of the most important determinants of vulnerability to adverse effects of water quality (Schaffelke et al., 2013). For example, reefs are likely to be more vulnerable if located close to river mouths (especially adjacent to disturbed land areas), in sheltered locations that reduce water exchange and where surrounded by areas of fine sediment prone to resuspension. In addition, reefs are typically in poorer condition and recover more slowly if they have been disturbed frequently, are not well connected to healthy source populations of broodstock for recruitment and have low abundances of herbivorous fish. New research on the effects of the various water quality parameters is discussed in detail below.

6.2.1 Light and sedimentation

Turbidity resulting from high concentrations of suspended sediment, and sedimentation, are two of the most widely recognised threats to coral reefs (Risk, 2014; Storlazzi et al., 2015). A number of assessments at the catchment and sub-catchment scales (Bartley et al., 2014; Lewis et al., 2015a; Lewis et al., 2015b) over the past few years have better identified and quantified the fine sediments on land from where they are transported into the marine environment (see also Chapter 2). Nutrient/organic-rich silt and clay-sized (<63 µm) sediment fractions are considered to pose the largest risk to coral reef organisms (Bartley et al., 2014). Recent reviews of the effects of various pressures on coral reefs have informed improved management of dredging operations (McCook et al., 2015; Jones, R. et al., 2016). The findings are equally applicable to the high turbidity and enhanced sedimentation following floods and wind-wave induced sediment resuspension.

Advances have been made recently in understanding the impacts of suspended sediments on the reproduction, recruitment and early life history of corals. High concentrations of suspended sediment can interfere with filter feeding, alter the quantity and quality of light available for photosynthesis by coral symbionts and smother corals with a fine layer of sediment that requires mucus production and energy to clear (Jones, R. et al., 2015b). Suspended sediments also impact the reproductive cycle and early life histories of corals (Jones, R. et al., 2015b). Two of the newly recognised mechanisms for the negative effects of suspended sediments are the entanglement and entrapment of coral sperm by sediment particles (Ricardo et al., 2015) and the ballasting of the buoyant egg-sperm bundles (Ricardo et al., 2016a). Both impacts reduce the fertilisation success of corals. While developing embryos and larvae tolerated exposure to suspended sediments by having mechanisms to remove particles (Ricardo et al., 2016b), the colonisation of reef surfaces by coral spat is threatened by the deposition of fine, terrigenous sediments (Perez et al., 2014; Jones, R. et al., 2015b).

Despite these physiological and ecological challenges of sediments and light levels to most corals, some taxa are able to tolerate and survive in highly turbid and muddy environments in the coastal Great Barrier Reef and can build calcareous reefs on unconsolidated sediments (Ryan et al., 2016a; Ryan et al., 2016b). Analyses of coral cores from reefs near the Palm Island group of the central inshore Great Barrier Reef showed that reef accretion at Pandora Reef and Havannah Island was consistent and rapid for the past 1000 years (Roff et al., 2015). Cores from nearby Pelorus Island, however, indicated dramatic changes in species composition starting in 1920, with few observations of the coral genus *Acropora* in death assemblages after 1980 (Roff et al., 2013). *Acropora* species are generally associated with reefs in clear water or at very shallow depth (Thompson et al., 2016), indicating a high light requirement. Roff et al. (2013) hypothesise that increases in sediment and nutrient loading following European settlement have resulted in a lack of recovery of *Acropora* assemblages following disturbance events (high water temperatures, tropical cyclones and flood events).

Analyses of dead massive *Porites* corals from the Palm Islands group, a genus generally tolerant to many environmental pressures, indicate two periods of recent mortality (Clark et al., 2014). The first period coincided with the 1997-1998 mass coral bleaching event, the second with significant flood events of the Burdekin River between 2007 and 2009 (Clark et al., 2014). If there is low recovery of these long-lived corals in the region, perhaps due to lack of broodstock connections to replenish these reefs, then *Porites* may well join *Acropora* in a local extirpation event.

In addition to effects of suspended sediments on corals, coral reef–associated herbivorous damselfish respond to suspended sediment. Their larval development, foraging success and habitat use is adversely affected by sediment concentrations that have been observed at Great Barrier Reef inshore reefs (Wenger et al., 2012; Johansen and Jones, 2013; Wenger and McCormick, 2013; Wenger et al., 2014). Grazing of macroalgae by herbivorous fishes is important on inshore or otherwise disturbed reefs (Cheal et al., 2012), as high macroalgae cover is an indicator of a loss of reef resilience (Thompson et al., 2016). However, herbivore diversity and activity are governed by complex ecological interactions, ranging from habitat and food type to predator presence (Rizzari et al., 2014).

6.2.2 Nutrients

Land run-off delivers a complex mixture of nutrients, suspended sediments and other compounds such as pesticides. Interactions between these water quality constituents are complex and often additive (Uthicke et al., 2016), which makes it challenging to separate the effects of nutrients from other co-occurring and correlated factors (Fabricius et al., 2013a).

The availability of excess dissolved inorganic nitrogen (DIN) alone has few direct adverse effects on corals (Fabricius, 2011). However, there is emerging evidence that under high temperatures, elevated DIN concentrations can increase the susceptibility of corals to bleaching (Vega Thurber et al., 2013; Wiedenmann et al., 2013; D'Angelo and Wiedenmann, 2014). Coral bleaching is ultimately a response to high reactive oxygen levels in the coral tissue produced by the corals' algal symbionts. This has been linked to high symbiont densities (Cunning and Baker, 2013) or changes at the cell-membrane level (D'Angelo and Wiedenmann, 2014), both of which can be influenced by excess nutrient availability or changes in nitrogen and phosphorus ratios (i.e. imbalances). Based on observations that reefs exposed to excess nitrogen were more susceptible to thermal bleaching and analyses of physiological measures such as coral thickness and symbiont density, Wooldridge (in press) proposes a further mechanism that exacerbates coral bleaching in nutrient-enriched waters: additional nutrient availability leads to increased zooxanthella densities but reduces energy available for the coral to build tissue reserves to resist stress events such as high temperatures. The interactions between water quality and heat stress are only beginning to be understood[28] with nutrient enrichment, low light and ocean acidification thought to play important roles (Cunning and Baker, 2013). Although there is still uncertainty about the interactions, Fabricius et al. (2013a) propose a conceptual framework that illustrates two important points: (i) nutrients and light can be either a stress or a beneficial factor, with optimum responses at species-specific tolerance levels and detrimental effects if rates are much higher or lower; (ii) 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 food preferences and trophic plasticity vary greatly between species. Fabricius et al. (2013a) conclude that in more eutrophic environments, as

[28] A National Environmental Science Programme's Tropical Water Quality Hub project 'Quantifying the linkages between water quality and the thermal tolerance of Great Barrier Reef coral reefs' commenced in 2017. nesptropical.edu.au/index.php/round-3-projects/project-3-3-1/

found in parts of the Great Barrier Reef after river floods (McKinnon et al., 2013), exposure to additional nutrients is predominantly a stress factor for most coral species. It is expected that improvement of water quality would improve the tolerance of inshore corals to thermal stress, although Hughes et al. (2017) suggest that good water quality may not add extra benefit during very extreme thermal stress events as in 2016.

The responses of early life history stages of the common inshore coral *Acropora tenuis* to a combination of excess inorganic and organic nutrients and elevated temperatures indicate that recruitment and recovery potential of this species may be limited at Great Barrier Reef inshore reefs due to poor water quality (Humanes et al., 2016).

As a direct effect, excess nutrient availability, especially of nitrogen, can promote the growth of fleshy macroalgae at locations with sufficient light (Schaffelke et al., 2005). Macroalgae are more abundant on reefs with high concentrations of water column chlorophyll, which is responsive to nutrient availability (De'ath and Fabricius, 2010). High macroalgal biomass on reefs has a number of adverse effects on corals: space competition (e.g. McCook et al., 2001), affecting coral metabolism by altering the corals' microenvironment (Hauri et al., 2010), reducing coral settlement through a number of mechanisms (Birrell et al., 2008) and increasing the susceptibility to coral disease (Morrow et al., 2012).

Coral disease is a significant cause of coral cover declines on the Great Barrier Reef (Osborne et al., 2011) and is predicted to worsen with global pressures of increasing temperature and ocean acidification (Maynard et al., 2015; O'Brien et al., 2016). While coral disease is considered a general stress response of corals, it has been positively correlated to sedimentation and elevated concentrations of nutrients and organic matter (Harvell et al., 2007; Haapkylä et al., 2011; Vega Thurber et al., 2013; Thompson et al., 2014; Pollock et al., 2016). Coral disease also appears to be exacerbated by physical injuries, such as those caused by tropical cyclones, derelict fishing gear (Williamson et al., 2015) and crown-of-thorns starfish outbreaks (Brandt et al., 2013; Katz et al., 2014; Lamb et al., 2016). Recent research has also identified a virus infection associated with bleached corals, a further indication of the importance of microbial agents in corals' stress responses (Correa et al., 2016).

6.2.3 Water quality and outbreaks of the crown-of-thorns starfish

Perhaps the most important indirect effect of excess nutrients on Great Barrier Reef coral reefs is the assumed link between high nitrogen availability and outbreaks of the coral-eating crown-of-thorns starfish, a native pest. When crown-of-thorns starfish occur in plague proportions they can significantly reduce the coral cover on a reef (De'ath et al., 2012). A new outbreak currently underway in the Cairns region has spread to reefs at the latitude of about Ingham and is expected to progress southward under the influence of regional southerly currents over the next few years.

The Great Barrier Reef is vulnerable to crown-of-thorns starfish outbreaks because of its size and connectivity between reefs (Hock et al., 2014), which allow outbreaks to spread via larval stages once primary outbreaks have been initiated. High nutrient availability, such as observed after a significant flood event, increases phytoplankton biomass, which is the food source of the planktonic crown-of-thorns starfish larvae. This in turn increases their survival, ultimately increasing the likelihood of crown-of-thorns starfish population outbreaks (Fabricius et al., 2010). The four crown-of-thorns starfish outbreak waves in the Great Barrier Reef since the 1960s all followed extreme floods of the Burdekin River and the rivers along the Wet Tropics coast, in particular the Russell–Mulgrave River (Furnas et al., 2013; Brinkman et al., 2014). However, the lag time between floods and outbreak detection varies. Past outbreaks that were sufficiently monitored followed a similar geographic pattern with outbreaks first being detected north of Cairns and subsequently spreading to the south (Vanhatalo et al., 2016). Crown-of-thorns starfish larvae south of Cairns were already detected in 2014 using genetic methods (Uthicke et al., 2015a).

Recent experiments confirm the dependence of crown-of-thorns starfish larval growth and development on phytoplankton concentrations (Uthicke et al., 2015b; Wolfe et al., 2015; Pratchett et al., 2017; Wolfe et al., 2017). Wolfe et al. (2015) and Pratchett et al. (2017) suggested that larval growth is at an optimum at intermediate phytoplankton cell numbers, with growth and development rates declining beyond that number. However, the high cell numbers used in these studies are unlikely to occur in the field, even under flood conditions (Devlin et al., 2013b). While the nutrient-limitation hypothesis is currently the best explanation for the occurrence of crown-of-thorns starfish primary outbreaks, other factors such as facilitation of larval development under higher temperatures (Uthicke et al., 2015b) may modulate the response. Sources of nutrients other than land run-off may also be important, such as shelf-break upwelling, but these are not yet well enough understood (Wooldridge and Brodie, 2015). It has recently been suggested that nutrient levels on mid-shelf reefs are sufficiently elevated during flood plumes to also promote the spread of secondary outbreaks (Brodie et al., 2017b).

An additional factor promoting crown-of-thorns starfish outbreaks appears to be the higher retention of the larvae in the area between Cairns and Lizard Island, where all primary outbreaks have been observed, during periods of reduced current velocities associated with neutral ENSO phases (Wooldridge and Brodie, 2015). Pratchett et al. (2014) suggested that primary outbreaks in the Far Northern Management Area (e.g. near Lizard Island) are initiated through slow population build-up, and subsequently nutrients imported through flood events promote larval survivorship and thus (southward) spread and secondary outbreaks.

Alternative hypotheses to explain outbreaks, such as reduced predation by overharvesting of animals that potentially consume crown-of-thorns starfish (e.g. fish, Triton shell), are still discussed in the literature (see review in Pratchett et al., 2014) and the subject of current research. Recent studies showed that benthic invertebrates can deter settlement of crown-of-thorns starfish larvae (Cowan et al., 2016a) and that damselfish can prey upon crown-of-thorns starfish larvae (Cowan et al., 2016b). A recent review (Cowan et al., 2017) identified 80 invertebrate and vertebrate predators and concluded that crown-of-thorns starfish predation may modulate outbreaks. A recent qualitative modelling exercise also suggested that models including both nutrient run-off and fishing perform best to explain crown-of-thorns starfish outbreak dynamics on the Great Barrier Reef (Babcock et al., 2016) and suggested that both nutrient run-off reduction and protected zones should be adopted as management strategies for crown-of-thorns starfish.[29]

6.2.4 Pesticides

Photosystem II–inhibiting (PSII) pesticides are regularly detected in the inshore Great Barrier Reef lagoon and during flood events may exceed regulatory guidelines and at concentrations known to affect marine organisms (e.g. Gallen et al., 2015).

A recent report synthesised the state of knowledge of the effects of herbicides and insecticides on coral reef organisms (Devlin et al., 2015b). As there has been no significant research effort on this topic since 2013, only a brief summary is provided. PSII herbicides reduce the efficiency of photosynthesis and, in the longer term, can cause damage to photosynthesis processes in the photo-symbionts of corals and foraminifera and in crustose coralline algae. For corals, this can reduce the energy acquisition by the host coral, which can lead to metabolic stress and reduced reproductive output. Fish exposed to insecticides have shown impaired liver function. However, the effects of chronic exposures to herbicides and insecticides on inshore coral reef organisms remain largely unknown. The assessment of the risks to Great Barrier Reef organisms of 'alternate' pesticides that are starting to be applied in the Great Barrier Reef region is currently hampered by available relevant

[29] nesptropical.edu.au/index.php/round-2-projects/project-2-1-1/

toxicity information (Kroon et al., 2015a). Generating improved knowledge of the ecotoxicity of alternate pesticides to freshwater and marine organisms in the Great Barrier Reef region is the subject of a National Environmental Science Programme's (NESP) Tropical Water Quality Hub project that commenced in 2017.[30]

Because pesticides, along with other pollutants, are predominantly transported into the Great Barrier Reef lagoon during summer flood events the interaction between compounds and with other environmental pressures is important and is the subject of recent and ongoing research (e.g. the effect of pesticide mixtures, the combination of pesticides with low salinity, low light, high suspended sediment concentrations, elevated nutrients and high temperatures). A first attempt to quantify the potential cumulative effects is presented in Lewis et al. (2013).

The highest concentrations of PSII pesticides can be expected during the warmest (summer) months. When simultaneously exposed to PSII pesticides and elevated temperatures, corals and foraminifera were more sensitive than normal to thermal stress and risk of bleaching (Negri et al., 2011; van Dam et al., 2012). The complex interactions among multiple stressors will often be unpredictable on the basis of knowledge of single effects. As the pollutant load to the Great Barrier Reef contains multiple chemicals, it is recommended to set load reduction targets for mixtures of pollutants, especially pesticides, using toxicity-based loads that consider the relative toxicity of each compound (Smith, J.N. et al., 2016; Smith, R.A. et al., 2016).

6.3 Ocean acidification

The results of recent experimental and field research into the effects of current and future ocean acidification on coral reefs provide strong evidence for adverse effects on the physiology and behaviour of many coral reef organisms, ranging from microbes to fish (Albright et al., 2016a; Sweet and Brown, 2016; Webster et al., 2016).

Ecosystem-wide effects are less well understood but field studies at sites with naturally high CO_2, mimicking projected future ocean acidification, show wide-ranging changes on coral reefs, for example in Papua New Guinea. At high-CO_2 sites, coral reef communities were dominated by tolerant but structurally less complex massive corals (with implications for the mega-diversity of coral reefs; Fabricius et al., 2013b), had lower abundance of crustose coralline algae (an important settlement substratum for coral larvae and binding element of the reef framework), but increased abundance of macroalgae and seagrasses (Fabricius et al., 2011). A recent field experiment on Heron Island in the southern Great Barrier Reef showed that ocean acidification may already be adversely affecting reef net calcification and, hence, growth (Albright et al., 2016b). Another documented effect of ocean acidification is increased bioerosion (Reyes-Nivia et al., 2013; Enochs et al., 2016), an effect which is exacerbated by high nutrient availability (Reyes-Nivia et al., 2013; DeCarlo et al., 2015). This is particularly relevant to Great Barrier Reef inshore reefs where higher incidence of bioerosion, which leads to a weakening of coral skeletons and the reef framework as a whole, has been associated with exposure to land run-off (Le Grand and Fabricius, 2010).

7. Resilience of Great Barrier Reef coastal and marine ecosystems

The environment in which Great Barrier Reef coastal and marine ecosystems exist is already changing, with ocean warming and ocean acidification predicted to intensify over the coming

[30] Project 3.1.5 'Ecotoxicology of pesticides on the Great Barrier Reef for guideline development and risk assessments'

decades (CSIRO and Bureau of Meteorology, 2015). Understanding the resilience[31] of ecosystems to environmental change is essential for management and conservation measures to maintain critical ecosystem functions and services (Anthony et al., 2015; Anthony, 2016).

7.1 Recovery potential of coastal and marine ecosystems

Because coastal wetlands and mangroves are not subject to systematic monitoring, there is limited information about their general resilience. At Hinchinbrook Island, after TC Yasi in 2011, damage was caused to 17% of the mangrove area but recovery of the damaged forest was observed after one year (Asbridge et al., 2015).

The capacity of seagrass meadows to recover community structure following disturbance depends on the interaction between light availability, nutrient loads, habitat properties and the availability of propagules to establish new populations (Grech et al., 2016; McKenzie et al., 2016b). While aspects of recovery in seagrass (e.g. changes in species composition) have been documented through monitoring, the recovery processes (e.g. triggers for seed germination, seed viability, seed bank thresholds, sediment conditions, species interactions) remain unknown; thus precluding accurate prediction of recovery rates for Great Barrier Reef seagrass communities. Current best understanding is that recovery of seagrass meadows is species-dependent and can take between two and eight years without further disturbances (McKenzie et al., 2016b). At present, Great Barrier Reef seagrass meadows continue to recover from losses experienced during the adverse weather of 2009-2011 and appear to have variable recovery potential due to historical factors including the availability of seed banks. Unfortunately, the recovery of some seagrass habitats in the Great Barrier Reef has been hindered by the simultaneous loss of ecosystem engineering benefits (e.g. sediment stabilisation) where negative feedbacks are undermining seagrass resilience (Adams et al., 2016). In such cases (e.g. south of Cairns to Hinchinbrook Island), seagrass recovery will be difficult without improvements in water quality. Understanding the influence of water quality on the feedback between sediment and light for seagrass ecosystem state (i.e. seagrass presence reduces suspended sediment to increase available light thereby favouring growth, whereas seagrass absence increases turbidity thereby reducing growth) will assist in maintaining long-term resilience of these important ecosystems as other cumulative impacts worsen due to changes in disturbance regimes.

Coral reefs are sensitive to disturbances such as tropical cyclones and crown-of-thorns starfish outbreaks and to environmental extremes such as high temperatures and major floods. The mid-shelf and outer shelf reefs of the Great Barrier Reef have shown capacity to recover from previous disturbances (see above section on condition of coral reefs). However, the regaining of pre-disturbance coral cover is not necessarily connected with the reassembly of the pre-disturbance community composition. Fast recovery and complete reassembly have been observed over a period of around 10 years on reefs with a high proportion of tabulate *Acropora* corals, while inshore reefs with other coral communities did not recover or reassemble their previous composition in that time frame (Johns et al., 2014). The importance of fast-growing tabulate corals for recovery of coral cover and three-dimensional reef structure is also emphasised by Ortiz et al. (2014). However, taxa with this growth form are generally more sensitive to a range of disturbances compared to other corals (e.g. Osborne et al., 2011; Berkelmans et al., 2012). On inshore reefs, the cumulative impact of tropical cyclones, extreme flood events and recent outbreaks of crown-of-thorns starfish resulted in declines in coral community condition during 2012-2014. Since 2014, indicators of coral reef resilience have improved, coinciding with a period of below average rainfall and, hence, low land run-off (Thompson et al., 2016). This demonstrates the inherent ability of communities to recover from acute disturbances when catchment loads are reduced. As a direct management tool, the

[31] The most widely used definition of ecological resilience considers two main dimensions: that of resistance to pressures and disturbance and the ability for recovery from impacts (Hodgson et al., 2015).

protection from fishing pressure in no-take zones has been associated with greater resistance to disturbance and shorter recovery times of Great Barrier Reef reefs, for both coral cover and reef fish community composition (Mellin et al., 2016).

7.2 Cumulative impacts and the challenge to managers

The cumulative impacts of global pressures, in concert with local disturbances (e.g. tropical cyclones) and local anthropogenic pressures such as land run-off, are likely to reduce the resilience of Great Barrier Reef costal and marine ecosystems (Anthony et al., 2015; Anthony, 2016). However, future conditions of coastal and marine ecosystems are difficult to predict as the responses to multiple pressures are only beginning to be understood, vary between organisms and studies and show complex spatio-temporal patterns, interactions and feedback loops (van Hooidonk et al., 2014; Sweet and Brown, 2016; Unsworth et al., 2015; Uthicke et al., 2016; van de Leemput et al., 2016; Harborne et al., 2017). While this knowledge is still incomplete for Great Barrier Reef ecosystems, insights from other terrestrial and aquatic ecosystems suggest that diversity and variability in responses to pressures are important components of resilience (Carpenter et al., 2015).

Lough and Cantin (2014) describe two types of responses of corals to warmer ocean temperatures, based on growth measurements in coral cores: (i) high temperature events leading to distinct, but perhaps intermittent, growth reductions, and (ii) a chronic growth decline as temperatures regularly exceed coral growth optima. For coral reefs to persist, it will be essential that future calcification and growth rates are sufficient to counteract biological and physical erosion processes, highlighting the importance of managing local pressures that may reduce growth rates. Similarly, two types of responses to warming water are expected for seagrasses: (i) high temperature at low tide inhibits photosynthesis and may induce mortality if sustained over many days (Campbell et al., 2006; Collier and Waycott 2014), and (ii) increases in respiration rate with chronic rises in water temperature can slow growth, particularly in meadows where photosynthetic rates are limited by water quality (low light, presence of herbicides; Collier et al., 2011; Collier et al., 2016a; Wilkinson et al., 2017). It is also essential to understand the capacity of organisms to acclimatise or adapt to their changing environment. While subject to current research efforts for corals and seagrass, the understanding of acclimatisation and adaptation rates is currently poor and debated (Logan et al., 2014; Palumbi et al., 2014; Sweet and Brown, 2016).

The key management approach in the Reef 2050 Long-Term Sustainability Plan (Commonwealth of Australia, 2015) to support ecosystem resilience has been the mitigation of local stressors, such as land-based sources of pollution and coastal development, and the management of direct uses, including fishing, to 'buy' time for coastal and marine organisms to adapt to the continued intensification of global pressures (Falkenberg et al., 2013; Kennedy et al., 2013; Kroon et al., 2014b; McClanahan et al., 2014; Kroon et al., 2016). A recent modelling study by Ortiz et al. (2014) indicates that ecosystem-level benefits of reduced carbon emissions (assuming the Representative Concentration Pathway 2.6; IPCC, 2014) would become noticeable in the Great Barrier Reef within 20–30 years, that is, within the duration of the Reef 2050 Plan. This means that local management options may be especially important during this period to maintain as much biodiversity, ecosystem function and services as possible into the future. However, the sustained implementation of local management strategies is recognised to be challenging and there is high uncertainty around the types of ecosystems and ecological functions that would persist if local pressures were sufficiently reduced or removed (van Hooidonk et al., 2016). Other analyses question whether global pressures may soon outweigh the positive effects of conventional local management (Anthony et al., 2015; Aswani et al., 2015; Albright et al., 2016a; Anthony 2016; van Hooidonk et al., 2016). The latter scenario may require additional, more aggressive local management strategies and/or intervention approaches to halt the ongoing and future degradation of coastal and marine ecosystems and maintain their ecological function, for example by active restoration of wetlands, seagrass meadows or coral reefs (see Chapter 4; van Oppen et al., 2015; van Katwijk et al., 2016; van Oppen et al., 2017).

8. Knowledge gaps and priority areas of further research

There is still appreciable uncertainty in our knowledge of the responses of coastal and marine ecosystems to the cumulative impacts of multiple pressures. Below are key knowledge gaps that have been identified in this review.

To improve assessments of coastal and marine water quality and the attribution of changes to catchment activities and river water quality, it is important to:

- improve validation and calibration of the eReefs biogeochemical model, through *in situ* observations of water quality from areas other than coastal Marine Monitoring Program locations (e.g. Cape York, mid- and offshore areas)
- clarify our understanding of the spatial and temporal patterns of water quality parameters (Chlorophyll-*a,* nutrient and carbon species*,* water clarity and benthic irradiance) in the Great Barrier Reef, for the purpose of refining existing water quality guideline values
- determine the transformation rates and processes and the bioavailability of terrestrially sourced particulate and organic nitrogen in the marine environment. These nitrogen forms dominate the terrestrially derived nitrogen and may play a critical roles in driving phytoplankton production and composition (which is important for coastal water clarity and food webs, including promoting and sustaining crown-of-thorns starfish outbreaks)
- determine the transport, fate and impacts of the finest sediment factions and the formation and ecological significance of organic flocs in the receiving marine environment
- inform future risk assessments by conducting targeted monitoring campaigns for contaminants that may pose a relatively high risk to coastal and marine ecosystems, including marine debris / microplastics (reef-wide), antifouling paint components (shipping lanes and anchoring areas) and personal care products (coastal and tourism sites)
- ensure that all water quality data are available in the public domain for improving determinations of marine baselines and environmental risk assessments.

To better understand and attribute responses of coastal and marine ecosystems to critical drivers of ecological changes, especially those linked to catchment use, temperature stress and ocean acidification, it is important to:

- identify the impacts of nutrients on the condition of freshwater floodplain wetlands and the tolerance thresholds and tipping points of their key species to multiple stressors;
- determine the areal extent of sediment-infilling of freshwater wetlands and identify the resulting loss of wetlands values;
- quantify tolerance thresholds and tipping points in key seagrass and coral reef species under exposure to single and multiple pressures from water quality, climate change and ocean and coastal acidification;
- improve understanding of recovery processes of seagrass (e.g. seed germination triggers, seed viability, seed bank thresholds, sediment conditions, species interactions) and the environmental factors affecting these that are needed to refine predictions of seagrass meadow resilience;
- improve understanding of reef recovery and coral community reassembly processes (fecundity, larval settlement cues, post-settlement survival, growth rates, coral–macroalgal balance) and the environmental factors affecting these (including water quality), to refine predictions of coral reef resilience;

- develop spatial ecosystem models to predict future ecosystem condition, resilience and recovery of key coastal and marine ecosystems, with an immediate focus on coral reefs and seagrass meadows;
- define desired states of seagrass meadows and coral reefs so that ecologically relevant targets can be set for water quality-related pressures such as light (i.e. long-term light requirements) and terrestrially sourced sediment, nutrient and pesticide loads;
- identify the relevant characteristics (e.g. recruitment, connectivity, amenable environmental conditions) and locations of 'sites/zones-of-hope' under different scenarios and settings, that is, areas to maximise return on investments into water-quality action, no-take zones, crown-of-thorns starfish control in the face of continued climate change and ocean acidification;
- improve understanding of the scope and rates of acclimatisation and adaptation of coral reef taxa and seagrasses to environmental change, in particular a changing climate;
- improve understanding of the mechanisms and processes by which nutrient run-off promotes crown-of-thorns starfish outbreaks to better focus catchment management investment that will mitigate crown-of-thorns starfish impacts. A specific need is to quantify phytoplankton responses in terms of suitability as crown-of-thorns starfish larval food (as energy or organic carbon content rather than chlorophyll) in response to various nutrient concentrations, forms and ratios; and
- prioritise freshwater and estuarine barriers for mitigation or removal to improve water quality and fish passage in coastal ecosystems.

9. Overall conclusions

Acknowledging that this chapter represents an update of the current state of knowledge since the last Scientific Consensus Statement (Brodie et al., 2013), the overall conclusions are summarised in a table below against the key findings from the 2013 Scientific Consensus Statement relevant to this chapter. In addition, the table contains a summary of the uncertainties in the available information and the knowledge gaps identified during the review.

Established knowledge and understanding (based on 2013 Scientific Consensus Statement findings—highlights)	New information in 2017	Unresolved or unknown areas (e.g. for further research)
• In 2010, a historically strong La Niña weather pattern developed, replacing an El Niño pattern. Between 2009 and 2012, seven cyclones affected north Queensland which produced substantial physical damage to shallow-water ecosystems and record flooding. Extreme rainfall in 2010-2011 and 2012-2013 resulted in extensive flood plumes along most of the coast and across much of the continental shelf in some regions.	• The 2013 water year (October to September) was the last in a sequence of very wet years (2007–2013) with total river discharge into the Great Barrier Reef of more than twice the long-term average. Total river discharge in the 2014-2016 water years was around or below average and no significant river flood events affected the Great Barrier Reef lagoon. • Several tropical cyclones affected the Great Barrier Reef between 2013 and 2015, with two severe cyclones in the area between Lockhart River and Cooktown in 2014 and 2015, respectively, and one severe cyclone crossing the Mackay/Capricorn Management Area. No cyclones were recorded in the Great Barrier Reef region in 2016. • Sea surface temperatures of the Great Barrier Reef have significantly warmed since the late 19th century by 0.80°C and will continue to warm. • Record warm sea surface temperatures were observed on the entire Great Barrier Reef in March, April and May 2016. Temperatures in the northern half of the Great Barrier Reef remained extremely high into late summer and autumn, while temperatures further south were slightly moderated	n/a

Established knowledge and understanding (based on 2013 Scientific Consensus Statement findings—highlights)	New information in 2017	Unresolved or unknown areas (e.g. for further research)
	in February 2016 by two tropical lows bringing cloud cover, rain and wind.	
• Suspended sediment discharges, especially after extreme weather events, negatively affect turbidity in inshore waters, reduce the light required by corals and seagrass meadows and increase the sedimentation of fine particles and organic-rich flocs (muddy marine snow) that can smother marine organisms.	• Monitoring time-series measurements for turbidity and suspended sediment concentrations in coastal waters of the Great Barrier Reef confirm strong spatial and temporal variability at individual reefs, and within and across regions. • Notwithstanding this high variability, recent analyses found that turbidity in the Great Barrier Reef lagoon is significantly affected by river flow and rainfall, irrespective of wave height, wave period and tidal range. • Acute disturbances such as severe tropical cyclones can result in localised sediment movement that is highly site-specific.	• Time series of benthic light, a more ecologically relevant indicator than—for example—turbidity, are still only available for a handful of locations.
• Significant new mangrove stands and landward range expansions in some areas of the Great Barrier Reef are correlated with increased sedimentation due to human activity. However, excessive sedimentation can reduce tree growth, bury seedlings and cause mortality. Increased productivity and growth in response to high nitrogen availability is offset by the increased probability of canopy loss and mortality during periods of drought or storm activity along gradients of increasing salinity. Remaining coastal wetlands are subject to sediment, nutrients and pesticides inputs from rainfall run-off and irrigation tailwater. These inputs and physical modifications to the wetlands contribute to loss of biodiversity and	• The extent of natural and near-natural wetlands has largely been maintained since the previous Scientific Consensus Statement at around 85% of pre-European extent, based on the most recent mapping assessment in 2013. • The extent of mapped freshwater swamps (palustrine wetlands) remains at 76% of pre-European extent, with an overall loss of 59 ha between 2009 and 2013. • Over 90% of pre-European extent of mangroves and saltmarshes remain in most catchments, except for the Barron, Mulgrave, Herbert, Bohle, Ross, Lower Burdekin, Pioneer, Calliope, Boyne, Fitzroy, Shoalwater, Burnett and Kolan catchments, with an overall loss of 293 ha between 2009 and 2013.	• Knowledge gaps remain around the impacts of nutrients on the condition of freshwater floodplain wetlands and the tolerance thresholds and tipping points of their key species to multiple stressors. • The areal extent of sediment-infilling of freshwater wetlands is unknown as are the resulting loss of wetlands values.

Established knowledge and understanding (based on 2013 Scientific Consensus Statement findings—highlights)	New information in 2017	Unresolved or unknown areas (e.g. for further research)
affect wetland structure and function, for example by facilitating weed growth, loss of connectivity between habitats, reduced oxygen levels and flow rate.		
• Inshore seagrass meadows along the developed Great Barrier Reef coast (i.e. south of Cooktown) have declined over the past three to five years and are in poor condition.	• Inshore seagrass meadows remain in poor condition, despite improvements in some seagrass condition indicators in some regions. There were overall improvements in abundance (above-ground per cent cover and biomass); however, reproductive effort declined, indicating a low capacity to recover from disturbances with the available seed resources.	• There are important knowledge gaps around the understanding of recovery processes of seagrass (e.g. seed germination triggers, seed viability, seed bank thresholds, sediment conditions, species interactions) and the environmental factors affecting these that are needed to refine predictions of seagrass meadow resilience.
• Recent loss of seagrass habitat as a result of severe weather events and degraded water quality has led to increased mortality of dugongs and green turtles.	• As seagrass abundance improved in many regions, in 2015, dugong mortalities decreased to the lowest levels since records commenced. In contrast, turtle mortalities have declined since 2011 but remain above the long-term median. The flow-on effects from seagrass loss to other associated fauna and/or multi-species fisheries are less obvious than in the case of the major herbivores; they may manifest as changes in community composition rather than losses of biomass.	• Ecosystem-wide effects are still poorly quantified and rely on observational data, which is often sparse.
• There is evidence of increases in seagrass leaf tissue nitrogen concentrations since 2005. Epiphyte loads that reduce light availability and impair seagrass growth have increased, possibly as a consequence of increased nutrient supply.	• Levels of leaf nitrogen and phosphorus have declined in inshore seagrasses since 2010, although carbon:nitrogen ratios of <20 indicate that nitrogen loads still exceed biological demand at coastal, reef and estuarine intertidal sites. There is no evidence that these excessive loads are directly detrimental to seagrass condition, although nutrient loading has been linked with the growth of epiphytes (fouling algae) on the leaves of seagrasses. At subtidal sites in the Wet Tropics and Burdekin regions there have been increased epiphyte burdens since 2014. Since	• There are significant knowledge gaps around the quantification of tolerance thresholds and tipping points in key seagrass species under exposure to single and multiple pressures from water quality, climate change and ocean and coastal acidification.

Established knowledge and understanding (based on 2013 Scientific Consensus Statement findings—highlights)	New information in 2017	Unresolved or unknown areas (e.g. for further research)
	2011, seagrass meadows in the Wet Tropics, Burdekin and Mackay Whitsunday regions have also endured high frequency of exposure to green, phytoplankton-rich water, indicating high bioavailability of nutrients. Epiphyte loads and exposure to phytoplankton-rich water interact to exacerbate light stress in seagrass meadows.	
• Many coastal and inshore seagrass meadows of the Great Barrier Reef are exposed to herbicide concentrations that adversely affect seagrass productivity. The contribution of herbicides to recent widespread seagrass losses is unknown.	• The half-lives of widely used PSII herbicides are more than 100 days, indicating high persistence and explaining their presence in the Great Barrier Reef year-round. • The PSII-inhibiting herbicide diuron, which is commonly found in Great Barrier Reef inshore waters, adversely affects seagrass photosynthetic efficiency at ambient concentrations in the Great Barrier Reef lagoon and at concentrations well below the Great Barrier Reef guideline. Furthermore, herbicides occur in mixtures and these have an additive effect on seagrass photosynthetic efficiency. • Seagrasses are also sensitive to chronic exposure of low herbicide concentrations, causing measurable impacts to photosynthetic carbon assimilation and energetic status.	• High concentrations of diuron and atrazine may have contributed to localised losses of seagrass meadows in the past, such as in Sarina Inlet in 2011. However, the overall contribution to widespread seagrass losses from 2009 to 2011 due to the cumulative impacts of herbicides and low benthic light from multiple years of above average rainfall and river discharge is unknown.
• Great Barrier Reef-wide coral cover has declined by approximately 50% since 1985, while coral cover on inshore reefs has declined by 34% since 2005. Coral cover in the northern Great Barrier Reef has remained stable. Causes of coral loss vary from reef to reef, depending on exposure to tropical cyclones, outbreaks of	• The condition of inshore reefs (Wet Tropics to Fitzroy region) has marginally improved between 2014 and 2015, but is still rated moderate to poor, depending on the region. The improvement coincided with a period of low disturbance, after a decline in condition during the 2007-2013 period of very wet years and tropical cyclone disturbances.	• There are significant knowledge gaps around the quantification of tolerance thresholds and tipping points in key seagrass and coral reef species under exposure to single and multiple pressures from water quality, climate change and ocean and coastal acidification and of the trajectories of reef recovery and coral community reassembly processes (fecundity, larval settlement cues,

Established knowledge and understanding (based on 2013 Scientific Consensus Statement findings—highlights)	**New information in 2017**	**Unresolved or unknown areas (e.g. for further research)**
crown-of-thorns starfish or coral disease, elevated temperatures causing coral bleaching and exposure to flood plumes.	• Hard coral cover on mid-shelf and offshore reefs south of around Cairns increased between 2012 and 2015, showing very fast recovery during a period of low disturbance. Coral cover declined in the Cairns/Cooktown Management Area, associated with the impact of tropical cyclones and crown-of-thorns starfish outbreaks. • A mass thermal coral bleaching event affected the Great Barrier Reef in 2016 and resulted in significant coral mortality, especially north of Port Douglas. The inshore reefs monitored as part of the Marine Monitoring Program (Wet Tropics to Fitzroy region) appear to have escaped major coral mortality.	post-settlement survival, growth rates, coral–macroalgal balance) and the environmental factors affecting these (including water quality).
• Evidence of the link between poor water quality, specifically nutrients, and crown-of-thorns starfish outbreaks has been greatly strengthened.	• The fourth crown-of-thorns starfish population outbreak commenced around 2010-2011 and has now progressed south, to reefs at the latitude of about Ingham. Currently, the most widely accepted hypothesis is that primary outbreaks are promoted by high nutrient availability, such as are observed after significant flood events, which increase larval food supply and survivorship. Recent research has supported this nutrient-limitation hypothesis, but some uncertainty about the exact chlorophyll-based threshold exists. There is some evidence for other factors contributing to outbreaks, such as removal of predators of adult crown-of-thorns starfish, retention of crown-of-thorns starfish larvae in the outbreak initiation region in the northern Great Barrier Reef and higher temperatures increasing larval survivorship.	• Several research studies are underway to investigate contributing factors to outbreaks, for example reduction of natural crown-of-thorns starfish predators (of all crown-of-thorns starfish life stages, that is, larvae, juveniles, adults). • Remaining knowledge gaps are around the understanding of the detailed mechanisms and processes by which nutrient run-off promotes crown-of-thorns starfish outbreaks. A specific need is to quantify phytoplankton responses in terms of suitability as crown-of-thorns starfish larval food (as energy or organic carbon content rather than chlorophyll) in response to various nutrient concentrations, forms and ratios.
• Poor water quality, especially elevated concentrations of and different ratios of	• Further evidence from experiments show that, in concert with increased temperature, high DIN	• Knowledge gaps are regarding which factors (nutrients, nutrient ratios, light/turbidity) are most important in

Established knowledge and understanding (based on 2013 Scientific Consensus Statement findings—highlights)	New information in 2017	Unresolved or unknown areas (e.g. for further research)
nutrients and high turbidity, has been shown to increase the likelihood of bleaching in corals.	concentrations and unbalanced nitrogen:phosphorus ratios can increase the susceptibility of corals to bleaching. • It is expected that improvement of water quality would improve the tolerance of inshore corals to thermal stress, although observations during the 2016 bleaching event suggest that good water quality may not add extra benefit during very extreme thermal stress events.	determining coral thermal tolerance, over which temperature and light ranges is water quality a determinant of variability in coral colony and reef thermal tolerance and if water quality affects the recovery potential of bleached corals.
• Pesticides pose a low to moderate risk to inshore coral reefs at current levels, but the consequences of long-term exposure at concentrations below those known to affect coral is not understood.	• PSII-inhibiting herbicides are regularly detected in the inshore Great Barrier Reef lagoon and during flood events may exceed regulatory guidelines and at concentrations known to affect marine organisms. • A recent report synthesised the state of knowledge of the effects of herbicides and insecticides on coral reef organisms; however, there has been no significant research effort on this topic since 2013.	• The assessment of the risks to Great Barrier Reef organisms of 'alternate' pesticides that are starting to be applied in the Great Barrier Reef region is currently hampered by available relevant toxicity information. Generating improved knowledge of the ecotoxicity of alternate pesticides to freshwater and marine organisms in the Great Barrier Reef region is the subject of a NESP Tropical Water Quality Hub project that commenced in 2017.
• The cumulative pressure of multiple stressors determines the state of marine ecosystems and the trajectory of recovery after disturbance. A 50% decline in Great Barrier Reef coral cover over the past three decades shows that the natural resilience of the ecosystem has been compromised by impacts from extreme rainfall and associated run-off from agriculture, thermal stress, salinity stress, light stress, cyclone damage and crown-of-thorns outbreaks. • The interactions of poor water quality with other pressures such as climate change are	• Research has improved the understanding of interactive and cumulative pressures as drivers of coastal and marine ecosystem condition and resilience, and of the interaction between ocean warming, acidification and tropical cyclones with local, anthropogenic pressures such as land run-off.	• The importance of interactive and cumulative pressures as drivers of ecosystem condition has been confirmed by a number of studies. However, future conditions of coastal and marine ecosystems are difficult to predict as the responses to multiple pressures are only beginning to be understood, vary between organisms and studies and show complex spatio-temporal patterns, interactions and feedback loops. To improve the knowledge, spatial ecosystem models, based on high-quality observational data, are required to predict future ecosystem condition, resilience and recovery of key coastal and marine ecosystems, with an immediate focus on coral reefs and seagrass meadows.

Established knowledge and understanding (based on 2013 Scientific Consensus Statement findings—highlights)	New information in 2017	Unresolved or unknown areas (e.g. for further research)
largely unknown, but could increase the risk to Great Barrier Reef ecosystems.		
• Reducing end-of-catchment loads of nutrients, sediments and pesticides will help enhance reef resilience in the face of continuing climate change pressures. For example, if the impacts of crown-of-thorns starfish were reduced following nitrogen load reduction from the Wet Tropics, coral cover is predicted to either recover or at least stabilise.	• Coral reefs are sensitive to disturbances such as cyclones and crown-of-thorns starfish outbreaks and to environmental extremes such as high temperatures and major floods. The recent period of low rainfall and run-off demonstrates the inherent ability of inshore reefs communities to recover from acute disturbances during periods of reduced catchment loads. The mid-shelf and outer shelf reefs of the Great Barrier Reef have shown capacity to rapidly recover from recent disturbances, mostly through increase in cover of fast-growing coral species.	• To improve assessments of coastal and marine water quality and the attribution of changes to catchment activities and river water quality, it is important to: — improve validation and calibration of the eReefs biogeochemical model, through *in situ* observations of water quality from areas other than coastal Marine Monitoring Program locations (e.g. Cape York, mid- and offshore areas). — clarify our understanding of the spatial and temporal patterns of water quality parameters in the Great Barrier Reef, for the purpose of refining existing water quality guideline values. — determine the bioavailability of terrestrially sourced particulate nitrogen in the marine environment; the nitrogen forms dominate the terrestrially derived nitrogen and may play a critical role in driving phytoplankton production and composition (which is important for coastal water clarity and food webs, including promoting and sustaining crown-of-thorns starfish outbreaks).

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