Water quality studies in the Great Barrier Reef region and links to management, 1890 – 2016

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with a foreword by Jane Waterhouse and Richard Pearson

Publication No. 21/01

January, 2021

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This document should be cited as:
Brodie, JE, 2021, Water quality studies in the Great Barrier Reef region and links to management, 1890 – 2016. Centre for Tropical Water & Aquatic Ecosystem Research (TropWATER) Publication 21/01, James Cook University, Townsville, 71 pp.

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This publication has been compiled by the Centre for Tropical Water & Aquatic Ecosystem Research (TropWATER), James Cook University.

Acknowledgements

Acknowledgements directly relevant to this publication recognise support and editorial comment by Richard Pearson, Jane Waterhouse, Stephen Lewis, Eric Wolanski and Zoe Bainbridge. Damien Burrows, Tricia Boyd and Karen Wood of TropWATER facilitated production of the current version.

Cover photo from Logan et al. (2013)

Contents

Foreword

This document was originally written by Jon Brodie as the Introduction to his PhD thesis[1]. The thesis was based on a series of his publications with a new Introduction and Discussion. The Discussion was published in 2016 (Brodie and Pearson 2016), but the Introduction remained unpublished. Jon was happy to publish it in a journal or as the current document, but sadly there was insufficient time before he died. We decided that in order to most closely retain the original, including over 450 references, we would follow the current course.

Both Jane and Richard have had a long association with Jon. Latterly, Richard supervised his PhD, while both of us commented on its content. We both contributed to very minor editing of this version of the Introduction, including removal of material relevant only to the PhD document itself.

Needless to say, we have both benefited tremendously from Jon's wisdom and friendship, and have long admired his advocacy, some of which appears or is cited in this document. We wish to acknowledge his immense contribution to water quality management in the Great Barrier Reef catchment, to improved management of the Great Barrier Reef, and to significantly raising the profile of the Reef and its catchment among government circles and the public alike. He was a sad loss to the community when he died in June 2020.

Richard Pearson (James Cook University, Townsville) • Jane Waterhouse (C_2O Consulting, Townsville)

[1]Brodie, JE, 2016. Terrestrial pollutant runoff to the Great Barrier Reef: effects, causes, sources and management. PhD thesis, James Cook University, Townsville, Queensland.

1. Introduction and background

This review outlines research and management into water quality in the The Great Barrier Reef (GBR) region, including the GBR proper and its catchment (Brodie and Pearson 2016). The science is reviewed until 2016, while an analysis of science to management ends at the formation of the Reef Water Quality Protection Plan (Reef Plan 2003) in 2003. However, debates regarding the resolution of scientific questions for management through to 2016 are discussed. An analysis of the success of management of the GBR since 2003, the current state of knowledge of the issue drawn from the 2013 Scientific Consensus Statement (Brodie et al. 2013a), including new knowledge gained in the period 2013 – 2016, a summary of the management plans being applied in 2016 and beyond, and assessment of the management steps needed to improve water quality in the GBR lagoon such that GBR water quality guidelines are largely met, and the likelihood of further substantive progress in managing the issue are discussed in Brodie and Pearson (2016).

The GBR Is an extensive coral reef system on the continental shelf of north-eastern Australia (Figure 1). The area of the official World Heritage Area (GBRWHA) is 348,000 km^2, with the Great Barrier Reef Marine Park (GBRMP) slightly smaller at 344,400 km^2 (Kenchington and Day 2011). Seven percent of the GBRWHA consists of 2900 individual coral reefs (Day, 2008, 2011). However, the GBR extends to the north beyond the boundary of the GBRWHA for more than 100 km, ending at Bramble Cay near Papua New Guinea at 9^0 S. The adjacent GBR catchment area (GBRCA) has an area of 424,000 km^2 (the Great Barrier Reef Catchment Area – GBRCA) (GBRMPA [Great Barrier Reef Marine Park Authority] 2012). The GBR has been managed as a national Marine Park since 1975 (*Great Barrier Reef Marine Park Act 1975*) with an ecosystem-based management approach and regular reporting framework (Dobbs et al. 2011), and was listed as a World Heritage Area in 1981 for its outstanding universal value (Day and Dobbs 2013). The GBRWHA encompasses large areas of coral reefs, seagrass meadows and mangrove forests, providing habitat to endangered and threatened marine megafauna including turtles, dugong, whales and dolphins (GBRMPA 2014) and high diversity of other plants and animals. In addition to its biodiversity and cultural heritage values, it is estimated that the collective monetary value of a broad range of services provided by the GBR is likely to be between $15 billion and $20 billion AUD per annum (Stoeckl et al. 2014). Several studies have described the scientific, legal and political background to the establishment of the GBRMP and the listing of the GBRWHA (Bowen and Bowen 2002; Laurence et al. 2002; Daley 2005, 2014).

Recognition of the potential impact of land degradation on downstream waters occurred even before the establishment of the GBRMP – for example, increased erosion from rainforest clearing (Douglas 1967). However, establishment of the GBRMP prompted greater focus on this issue, indicated by the new Great Barrier Reef Marine Park Authority (GBRMPA) employing staff to investigate such issues (Bennell 1979).

The idea that the GBR had a definable catchment came rather late in the planning process for the Marine Park but was well conceived by the late 1980s, when funding for end-of-river monitoring for sediment and nutrient loads was provided to M. Furnas and A. Mitchell at the Australian Institute of Marine Science (AIMS) by GBRMPA (Mitchell et al. 1991; Furnas 2003) and maps of the GBRCA started to be produced (e.g., Moss et al. 1992). The concept that, in managing water quality in the GBR, most management would have to take place on the GBRCA, often hundreds of kilometres inland on agricultural and urban lands, was also clarified by 1990 (e.g., Hunter and Rayment 1991; Brodie 1991).

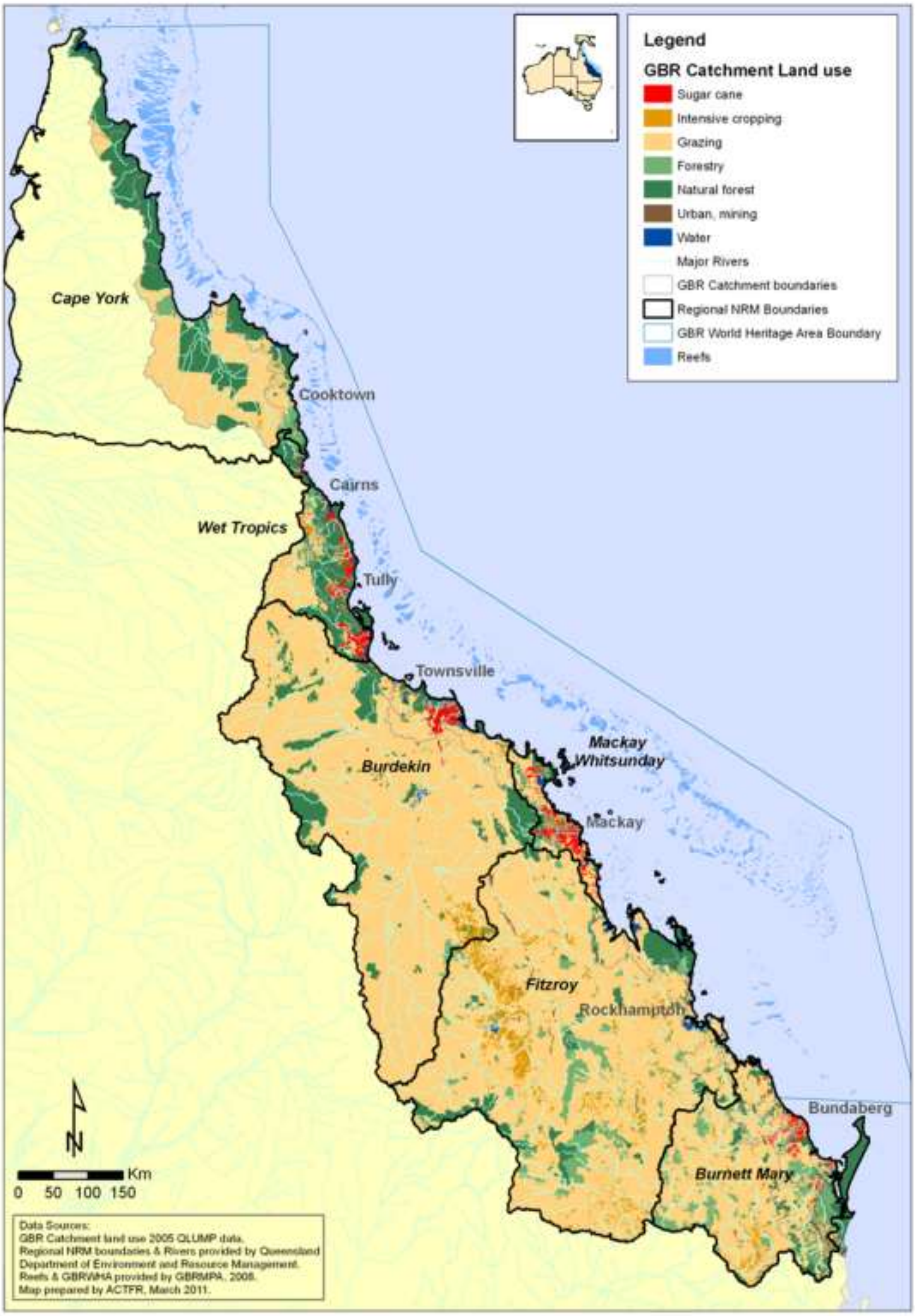

Figure 1. The Great Barrier Reef and its catchments, showing land uses and the six Natural Resource Management regions.

2. The historic context

The results of studies on the GBR carried out long before the establishment of the GBRMP, although not specifically addressing terrestrial runoff, are still used to examine "baseline" conditions in some areas of the GBR. Important examples of these studies include:

i. William Saville-Kent's photographs of reef flats, generally at low tide so the coral can be seen, in many areas of the GBR including the Torres Strait (Saville-Kent 1893). These photographs have been used frequently in the last 30 years to compare with current reef coral status (Bell 1992; Wachenfeld 1997; Hughes et al. 2010; Ryan et al. 2016; **Clark et al. 2016**). It is notable that Saville-Kent took the photographs in the 1890s specifically as a "baseline" so that future researchers could use them to assess change in the reefs. Other photographs of reef flats at low tide, dating from throughout the 20th century, have been used in a similar way (Endean 1982, Bell 1992, Wachenfeld 1997).

ii. Studies on Low Isles by the British Museum Expedition of 1928-29 were the then highpoint of scientific research on the GBR (Yonge 1930). Results associated with "water quality" parameters included the status of nutrients (Orr 1933), plankton (Marshall 1933) and reef-flat coral sedimentation (Marshall and Orr 1931). Results of these Low Isles studies have been used as starting points for more modern comparative research aimed at measuring change (Bell and Elmetri 1993; Stephenson et al. 1958).

   Subsequent studies focussed on the damaging effects of the 1934 cyclone on *Porites* sp. corals examined by Moorhouse (1936) and other changes to the island and reef (Spender 1936). Descriptions of the geomorphology of the area were published later (Fairbridge and Teichert 1947, 1948). Another cyclone in 1950 caused damage to the island's coral, and an expedition visited Low Isles in 1954 to assess the extent of this damage and of changes to the island and its biota since 1929. Results focussed on geomorphology and corals showed that the cyclone had caused great destruction to branching corals but that massive corals had, in most cases, survived (Stephenson and Wells 1956; Stephenson et al. 1958). Soft corals, which appeared to have been unaffected by the cyclone, had spread, and appeared to be competing with hard corals for the available substratum. The mangrove area had increased in extent compared to 1928.

iii. The Second International Coral Reef Society (ICRS) expedition was conducted by the GBR Committee on the M.V. Marco Polo from 22nd June to 2nd July 1973. The symposium was attended by 264 scientists and formal sessions were conducted while the ship was under way in order to allow the maximum time for the field program. During the symposium, 120 papers were presented. Water quality studies were conducted and several seminal papers were given at the Symposium – for example, on fertilisation studies at One Tree Island, where corals were subjected to several added nitrogen and phosphorus compounds for an extended period (Kinsey and Domm 1974; Kinsey and Davies 1979).

Commencing in the 1970s, GBRMPA initiated a number of workshops and technical reviews of water quality issues for the GBR. Some of the workshops were not specifically focused on water quality but were important for the light they shed on aspects of water quality dynamics – for example, the Fringing Reef Workshop (Baldwin 1987).

In 1984 GBRMPA organised a workshop that concentrated on heavy metals, polychlorinated biphenyls (PCBs) and other organochlorines (e.g., insecticides) and hydrocarbons in the GBRMP (Dutton 1985). In attempting to assign priorities to areas of further research, participants noted that sediments and nutrients were more likely to be of greater concern to the GBR than the three contaminant groups considered at that workshop. In particular, an area recommended for further research was: "the effects of agricultural fertilisers and other nutrients exported to the GBR from the mainland."

Important workshops, technical reports, reviews and strategic direction and policy statements in the 1980s, (summarised in Table 1) led to the identification by 1990 of sediment and nutrient discharged from the land, primarily from agriculture and urban development, as posing the greatest water quality threat to the GBR. In contrast pesticides (organochlorine insecticides) (Smillie and Waid 1985), hydrocarbons (Coates et al. 1986; Smith et al. 1985), heavy metals (Denton and Burdon-Jones 1986a,b), PCBs, sewage effluent bacterial status, anti-fouling toxins and oxygen-depleting substances were assessed as being of much lower threat (Table 1).

Following 1990 and the commencement of a formal water quality research and monitoring program, workshops were held and numerous reports, papers, reviews, strategic documents, policy statements and synthesis reports related to water quality were produced, many contributing to the first Scientific Consensus Statement (Williams et al. 2002), the Great Barrier Reef Catchment Water Quality Action Plan (Brodie et al. 2001a) and also thus to Reef Plan 2003 (Table 2). The “consensus” position derived from this body of work (see Section 1.8) formed the basis of water quality management after 2003.

3. Debates

During the 1980s, scientists and environmental managers engaged in a series of informal debates regarding issues relevant to GBR water quality, revealing gaps in understanding that helped to shape the research agenda in the 1990s and beyond. The debates on these issues have continued to the present and, although some “consensus” has developed from them, new research results open up the issues to further interpretation and debate as outlined below.

3.1 Is the GBR degraded by anthropogenic influence to any extent?

While there has been some consensus among scientists and managers that there is degradation due to human activities such as fishing, water quality impairment, tourism, recreation and climate change since the 1990s (Pandolfi et al. 2003; DeVantier et al. 2006; Bruno and Selig 2007; De’ath et al. 2012; Hughes et al. 2003; Hughes et al. 2011, 2015; Talbot 2000), the extent and severity of the level of degradation and its causes have been hotly disputed (Hopley 1988, 1989; Carter 2006; Ridd 2007; Ridd et al. 2011, 2013). In addition, there are some who have disputed whether any level of degradation has occurred and assert that the GBR is still in good condition (Larcombe and Woolfe 1999b; Starck 2005).

Table 1. Summary of workshops, technical reports, reviews and statements relevant to GBR water quality, 1983 – 1990.

Date	Editors/author	Title and details	Example paper
1983	Barnes D.J. (ed)	Perspectives on Coral Reefs 1983. Australian Institute of Marine Science, Townsville.	Crossland C.J., 1983. Dissolved nutrients in coral reef waters.
1983	Baker, J., Carter, R., Sammarco, P., Stark, K. (eds)	Proceedings of the inaugural Great Barrier Reef Conference: 1983. James Cook University and the Australian Institute of Marine Science. 545 pp.	Johannes, R., Wiebe, W., Crossland, C. 1984. Three patterns of nutrient flux in a coral reef community.
1984	Dutton, I. (ed)	Workshop on Contaminants in Waters of the Great Barrier Reef. Workshop Series No. 5. Great Barrier Reef Marine Park Authority, Townsville, 43 pp.	Burdon-Jones, C., Denton, G. 1984. Metals in marine organisms from the Great Barrier Reef – Study Outline 1980 – 1983.
1986	Dutton, I.M. (ed)	Workshop on the offshore effects of cyclone Winifred	Furnas, M. and Mitchell, A., 1986. Oceanographic conditions on the north Queensland shelf after passage of cyclone Winifred.
1987	Baldwin, C.L. (ed)	Fringing Reef Workshop: Science, Industry and Management. Workshop Series No. 9, Great Barrier Reef Marine Park Authority, Townsville, 280 pp.	Kinsey, D.W. 1987. Effects of run-off, siltation, and sewage.
1987	Baldwin, C.L. (ed)	Nutrients in the Great Barrier Reef Region 1987 Workshop Series No. 10.	Mitchell, A. 1987. River inputs of nutrients.
1987	Bell P., P. Greenfield, D. Connell & D. Hawker	Guidelines for management of waste discharges into the Great Barrier Reef Marine Park, Vol.1. Report to GBRMPA. 82pp.	
1990	Baldwin, C.	Impact of elevated nutrients in the Great Barrier Reef, GBRMPA Research Publication No. 20, GBRMPA, Townsville.	
1990	Yellowlees, D. (ed)	Conference on Land use patterns and nutrient loading of the Great Barrier Reef region James Cook University, Townsville.	Furnas, M. 1991. Nutrient status and trends in waters of the Great Barrier Reef Kinsey, D. W. 1991b. Water quality and its effect on reef ecology

Table 2. Important workshops, statements and reports relevant to GBR water quality, 1992-2004.

Date	Editors/authors	Title and details	Example paper
1992	Moss, A.J., Rayment, G.E., Reilly, N., Best, E.K.	A preliminary assessment of sediment and nutrient exports from Queensland coastal catchments, Technical Report No. 4, Queensland Department of Environment and Heritage, Brisbane, 33 p.	
1992	Brodie, J.	Sewage Policy for outfalls into the GBRMP. First established 1992, updated in 2005.	Brodie, J.E. 1991, 1994; Waterhouse, J. and Johnson, J. 2002.
1992	Hunter, H.M.	Agricultural contaminants in aquatic environments: a review. Department of Primary Industry, Brisbane.	Hunter 1992.
1994	Raymond, K. and Craik, W. (compilers).	A 25 year Strategic Plan for the Great Barrier Reef World Heritage Area, Great Barrier Reef Marine Park Authority, Townsville, 64 p.	Great Barrier Reef Marine Park Authority, 1994; Raymond 1996
1995	Wachelfeld, D., Oliver, J., Davis, K. (eds)	Proceedings of the State of the Great Barrier Reef World Heritage Area Workshop: proceedings of a technical workshop (Workshop No. 23) held in Townsville, Queensland, Australia, 27-29 November 1995. 561 pp.	Brodie, J. 1995b. The water quality status of the Great Barrier Reef World Heritage Area.
1995	Larcombe, P., Woolfe, K. (eds)	Great Barrier Reef: Terrigenous Sediment Flux and Human Impacts. 2nd ed. CRC Reef Research Centre.174 p	Larcombe et al. 1996. Terrigenous sediment fluxes and the central Great Barrier Reef shelf: the current state of knowledge.
1995	Zann, L., Kailola, P. (eds)	The State of the Marine Environment Report for Australia Technical Annex 2: Pollution. Great Barrier Reef Marine Park Authority, Townsville.	Brodie, J. 1995c. The problems of nutrients and eutrophication in the Australian marine environment.
1995	Furnas, M.J., Mitchell, A.W. & Skuza, M.	Nitrogen and phosphorus budgets for the central Great Barrier Reef shelf, GBRMPA Research Publication No. 36, GBRMPA, Townsville, 194 p.	Furnas et al. 1995
1996	Pulsford, J.S.	Historical Nutrient Usage in Coastal Queensland River Catchments Adjacent to the Great Barrier Reef Marine Park, Report to the Great Barrier Reef Marine Park Authority, Townsville.	Pulsford 1996
1996	H.M. Hunter, Eyres, A.G., Rayment, G.E. (eds).	Conference on Downstream Effects of Land Use. Queensland Department of Natural Resources, Brisbane	Neil, D.T., Yu, B. 1996. Fluvial sediment yield to the Great Barrier Reef lagoon: Spatial patterns and the effect of land use.
1996	Turia, N., Dalliston, C. (Compilers)	The Great Barrier Reef, science, use and management a national conference: proceedings. Vol. 1 (402 pp) and 2 (229 pp).	Rayment, G.E., Neil, D.T. 1996. Sources of material in river discharge.
1997	Steven, A. (ed)	Cyclone Sadie Flood Plumes in the Great Barrier Reef Lagoon: Composition and Consequences. Workshop Series no. 22, Great Barrier Reef Marine Park Authority, Townsville.	Mitchell and Bramley 1997. Brodie 1995a.

Date	Editors/authors	Title and details	Example paper
1997	Cosser, P.R. (ed)	Australia: State of the Environment Technical Paper Series (Estuaries and the Sea) (Nutrients in marine and estuarine environments, Technical Paper Series (Estuaries and the Sea), Department of the Environment, Canberra.).	Brodie, J. 1997. Nutrients in the Great Barrier Reef Region.
2000	Haynes, D. et al.	Great Barrier Reef water quality: Current issues, Great Barrier Reef Marine Park Authority, Townsville.	Haynes et al. 2001
2000	Gilbert, M., Brodie, J.	Population and major land use in the Great Barrier Reef catchment area: spatial and temporal trends. GBRMPA Research Publication Series, Great Barrier Reef Marine Park Authority, Townsville.	Gilbert and Brodie 2001
2001	Prosser, IP, Rustomji P. Young WJ, Moran CJ, Hughes AO	Constructing river basin sediment budgets for the National Land and Water Resource Audit. Technical Report 15/01, CSIRO Land and Water, Canberra.	
2003	Baker, J. (ed)	A report on the study of land-sourced pollutants and their impacts on water quality in an adjacent to the Great Barrier Reef: An assessment to guide the development of management plans to halt any decline in the water quality of river catchments draining to the Reef, as a result of land-based pollution, and to achieve the long-term goal of reversing any trend in declining water quality.	Baker 2003
2001	Wolanski, E.	Oceanographic processes of coral reefs: physical and biological links in the Great Barrier Reef ISBN 084930833Z, 376 pages	
2001	Devlin, M., Waterhouse, J., Taylor, J., Brodie, J.	Flood plumes in the Great Barrier Reef: spatial and temporal patterns in composition and distribution. GBRMPA Research Publication Series, Great Barrier Reef Marine Park Authority, Townsville.	Devlin et al. 2001
2003	Productivity Commission	Industries, Land Use and Water Quality in the Great Barrier Reef Catchment, Research Report, Productivity Commission, Canberra.	
2003	Furnas, M.J.	Catchments and Corals: Terrestrial Runoff to the Great Barrier Reef. Australian Institute of Marine Science, CRC Reef. Townsville, Australia.	Furnas 2003
2004	Haynes D, Schaffelke B (eds).	Catchment to Reef. Water Quality Issues in the Great Barrier Reef Region. 9-11 March 2004, Townsville. Conference Abstracts. CRC Reef Research Centre Technical Report No. 53. CRC Reef Research Centre, Townsville.	Fabricius et al. 2005

One of the most visible signs of coral reef degradation has been where coral reef flats with good coral communities, photographed long ago, have been re-photographed more recently and found to have severely deteriorated. The most noteworthy of these reefs is at Stone Island in northern Edgecumbe Bay near Bowen, where cyclonic floods are believed to have caused extensive coral mortality in the area in 1918 (Hedley 1925; Rainford 1025). The reef flat was photographed in the 1890s (Saville-Kent 1893) and then again in 1994 (Wachenfeld 1997; Clark et al. 2016) (Figure 2). Wachenfeld mentioned several caveats in the use of such images, in particular that it was very difficult to get the same location, state of tide and perspective when taking the modern photograph. The photographs from Stone Island reef have been used to claim anthropogenic reef degradation (Bell 1992; Hughes et al. 2010) while others claim the differences in the photographs are due to different states in an ongoing intermittent disturbance regime (DeVantier et al. 1998). Wachenfeld (1997) indicated that out of 14 reefs investigated, six showed no obvious changes, four showed decreases in hard coral cover and four showed no obvious changes in some areas, but decreases in coral cover in others. While the study demonstrated that some reef flats had undergone significant decline in coral cover and diversity, the cause of the change could not be determined from the photographs alone.

In 2015 Piers Larcombe reignited debate as to the significance of the historical photos at Stone Island and Bramston Reefs just as new papers resulting from greatly improved dating techniques (Clark et al. 2012, 2014) and years of research at reefs in Edgecumbe Bay were being published (Ryan et al. 2016a,b; Clark et al. 2016). While Larcombe implied that coral growth has recovered at the reefs in this area in public presentations (with new photos), in fact coral recovery has only visibly occurred at Bramston Reef. There is good evidence that there were extensive coral-dominated reef flats at Stone Island in 1894 and 1915 that were not observable in 1994 or any time since. Recent surveys show there is still little coral on reef flats anywhere on Stone Island (Clark et al. 2016; Ryan et al. 2016a). In contrast, at Bramston Reef (a reef much closer to the coast), while there was extensive live coral on the reef flat in the 1890s, there was little live cover in 1994 (although dead massive corals were common), but there has been some recovery since (Ryan et al. 2016a; Clarke et al. 2016).

Debate has also occurred about the coral communities present on Low Isles in 1928 (Yonge 1930; Stephenson 1930, 1958) compared to recent times, with contrasting claims of change/degradation (Bell 1992) or no change (Frank and Jell 2006), with reef conditions shown to be at least equal to those at Heron Island at the southern end of the GBR through comparison of foram communities (Schueth and Frank 2008). Similarly, the water quality and plankton community data at Low Isles from 1928 has been compared to modern measurements, using both modern techniques and the original methods (Bell and Elmetri 1993). Secchi depth measurements show a halving of clarity between 1928 and 1997, attributed to greatly increased discharge of mud from rivers in the region (Wolanski and Spagnol 2000). Comparison of *Trichodesmium* abundance shows an approximate tenfold increase in the years between 1928 and 1992/93, attributed to increased delivery of phosphorus and other nutrients from intensely developed GBR catchments (Bell et al. 1999).

A more recent debate centres on the decline in calcification rate and its possible causes, such as bleaching, warmer water, ocean acidification, river discharge and reduced water quality (De'ath et al. 2009; Cooper et al. 2008; D'Olivo et al. 2013). Some researchers have attempted to refute the notion of wide-scale reduced calcification – "Have coral calcification rates slowed in the last twenty years?" (Ridd et al. 2013). De'ath et al. (2013) responded with "Yes: Coral calcification rates have

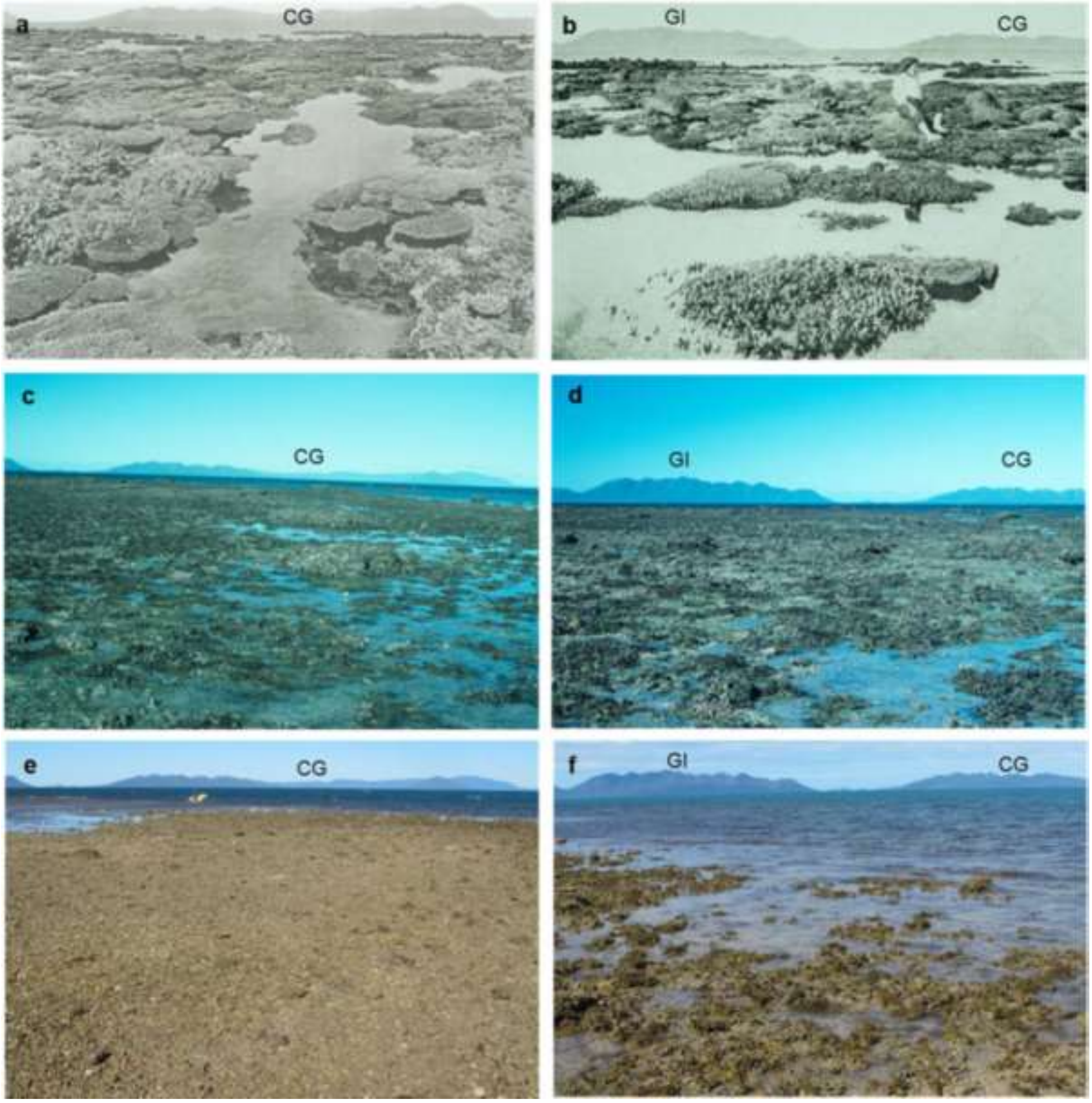

Figure 2. Historical and modern photographs of Stone Island. (a) Photograph taken by William Saville-Kent (1893) in ca. 1890 depicting high cover of branching and tabular *Acropora*; (b) Photograph taken in 1915 showing similarly high coral cover with large faviid colonies in the background of the image (photographer unknown); (c-d) Photographs of Stone Island taken in 1994 in conjunction with David Wachenfeld [Photographer: A. Elliot (© Commonwealth of Australia GBRMPA)]; (e) Photograph taken during this study on 30 July 2012 representing a time-series for images a and c (Photographer: T. Clark); (f) Photograph taken during this study on 31 July 2012 representing a time-series for images b and d (Photographer: H. Markham). Geological features in the background of the images used to identify the location of the historical photographs include Gloucester Island (GI) and Cape Gloucester (CG). Modified from Clark et al. (2016) with permissions from Dr Tara Clark and Ms Hannah Markham.

decreased in the last twenty-five years!" Meanwhile, Cantin and Lough (2014) showed that calcification rates can recover within 4 years following bleaching events. The cause of any reduced calcification is also in some dispute with thermal stress, bleaching and ocean acidification implicated in some studies (De'ath et al. 2009) while ocean acidification is seen as less of an issue in other studies (Cooper et al. 2008). Differences in causes depending on location across the shelf have been postulated, with inner-shelf reefs showing decreased calcification due to thermal stress (Cooper et al. 2008) or water quality issues (D'Olivo et al. 2013), including pH changes due to terrestrial runoff (D'Olivo et al. 2015), whereas reduced calcification on mid- and outer-shelf reefs is attributed to thermal stress (D'Olivo et al. 2013).

A comprehensive assessment of the current status of GBR coral reefs based on all the available monitoring data has not been produced. The landmark De'ath et al. (2012) paper, which documented a 50% decline in coral cover (from about 28% to 14%) on the reefs surveyed in the AIMS (2016) Long Term Monitoring Program (LTMP), used reefs that were mostly situated mid-shelf on the GBR, with only a few on the outer or inner shelves, and relied on manta-tow data from shallow sections of the reefs. For inner-shelf reefs the results from the Marine Monitoring Program (MMP) (which specifically targets inner-shelf reefs) were summarised in Thompson et al. (2014), as discussed below. Osborn et al (2011) used results from video transects at about 6 - 8m from LTMP reefs and ascribed somewhat different causes from those derived from the manta-tow results at the same reefs (De'ath et al. 2012). In addition, long-term studies of individual reefs have shown coral decline and recovery (e.g., Pandora Reef – Done et al. 2007). The likely decline in coral cover since the 1960s is approximately 75% (Bruno and Selig 2007; Hughes et al. 2011) from around 50% cover in the 1960s to the current 14%. Coral cover in the Torres Strait section of the GBR was in better condition than most of the rest of the GBR, averaging about 50% in 2014/15 (Sweatman et al. 2015); however, cover declined precipitously in 2016 with the major bleaching event (Normile 2016) here and across much of the northern GBR (GBRMPA 2016). Studies using coral cores (Lewis et al 2012a; Mallela et al 2013) and reef matrix cores (Roff et al 2013) have provided clear evidence for regional and local anthropogenic influences on reef growth, decline and recovery. A recent summary of reef condition and change in condition in the central GBR for the Burdekin WQIP (Coppo and Brodie, 2015), drawing on the LTMP (Australian Institute of marine Science 2015) and MMP (Thompson et al. 2014) surveys as well as some of the individual studies cited above, shows that reefs in this region have poor coral cover (<10%) with the only exceptions being Pandora and Middle Reefs, which are close to the coast.

Many questions remain as to the main factors driving GBR degradation but there is little doubt that the combined stresses of water quality decline and climate change impacts such as bleaching (Thompson and Dolman 2010; Done et al. 2007; D'Olivo et al. 2013) are important. While the debate continues, large-scale assessments of the "health" of the GBR continue – for example, GBRMPA's Outlook Reporting (GBRMPA 2009, 2014) – and management proceeds on the basis of a consensus view that the GBR is in poor condition due to combined stressors, that the decline in condition continues (GBRMPA 2014) and that a large increase in the management response is needed to reverse the decline (Hughes et al. 2015).

3.2 Debate over the water quality status, the extent and degree of damage from terrestrial pollutants, and the extent of eutrophication of the GBR

The debate about the water quality status of the GBR began in the 1980s (e.g., Kinsey 1988) and focussed largely on nutrients and the possibility that the GBR was, or might become, eutrophic. The paper that sparked the debate was Bell and Gabric (1991), which appeared in the CSIRO magazine *Search* with the title "Must GBR pollution become chronic before management reacts?" Four other papers were published on this question: Kinsey's "Can we resolve the nutrient issue for the reef" (Kinsey 1991a); Walker's "Is the reef really suffering from chronic pollution?" (Walker 1991); Barnes and Lough's "Nutrients and the need for scientific debate" (Barnes and Lough 1991); and Hopley's general response to the issues (Hopley 1991).

Given the title of Bell and Gabric's (1991) *Search* article it is now regrettable to be able to give a definitive "yes" in answer to their question. Indeed, pollution of the GBR did become chronic before important catchment management works were instituted under the Reef Rescue initiative in 2009 (Brodie et al. 2012a). Additionally, it is now widely believed that the time lag to management of 18 years meant that the works, when finally rolled out in catchments, were "too little, too late" (Brodie and Waterhouse 2012).

The initial debate involved claims that the GBR was already eutrophic (Bell and Gabric 1990, 1991; Bell 1991, 1992, 1993; Gabric and Bell 1993) and a range of other opinions that it was not (Walker 1991) or that it was not possible to decide with the data available around 1991 (Kinsey 1991; Barnes and Lough 1991; Hopley 1991). In subsequent years, various studies showed elevated concentrations of particulate and dissolved inorganic nutrients and chlorophyll-a in flood plumes (Brodie 1995, 1996; Brodie and Furnas 1996; Devlin et al. 2001; Devlin and Brodie 2005) while lower concentrations were observed in non-discharge periods (Furnas and Brodie 1996; Brodie et al. 1997). Chlorophyll concentrations, an indicator of phytoplankton biomass, were monitored extensively using grab sampling from the late 1980s to the mid-2000s with similar results – high concentrations in flood plumes (Devlin et al. 2001; Devlin and Brodie 2005) and lower concentrations in non-discharge conditions (Brodie et al. 2007), but with generally higher concentrations near the coast than further offshore, and higher concentrations south of Cooktown than on Cape York (Brodie et al. 1997, 2007). Information from remote sensing of chlorophyll concentrations using the Coastal Zone Colour Scanner (CZCS) suggested some degree of eutrophication in the GBR lagoon (Gabric et al. 1990) but, as for all satellite remote-sensing methods, there are doubts as to the reliability of chlorophyll measurements from satellite data in the turbid inshore waters of the GBR (Waterhouse and Brodie 2015). Bell and coauthors continued to publish on the state of eutrophication of the GBR (Bell et al. 2007, 2012).

The debate re-opened in 2014 with Bell et al. (2014a) asserting that "*it was now widely accepted that the lack of recovery of GBR reefs and the proliferation of COTS* [crown-of-thorns starfish] *are largely attributable to eutrophication. Evidence is emerging that coral skeletal disease (CSD) and coral bleaching are also promoted by eutrophication. Much of the increased fertility/eutrophication is due to the increased loads of nutrients exported via discharges from coastal developments*." Furnas et al. (2014) were highly critical of this paper, demonstrating that Bell et al. (2014a) misrepresent the current state of knowledge regarding water quality within the GBR and the processes that influence it in ten separate areas of knowledge, and also noted the lack of data available to validate Bell et al.'s assertions. Bell et al. (2014b) countered, claiming that "*while Furnas et al. (2014) suggest that* *there is a lack of relevant data to support some of the assertions/hypotheses there is sufficient data to draw robust conclusions. Bell et al. (2014b) agree that more data should be collected to better assess the impacts of STP discharges and links between eutrophication and the proliferation of the COTS larvae, jellyfish, diazotrophs, and CSD precursors. The current GBR research/monitoring programs are incapable of doing this and we recommend that a series of regular cross-shelf water-quality/ecological monitoring programs be established to collect the required data".*

Other studies focussed on experimental studies of the effects of nutrients on coral reefs. In the ENCORE experiment (Koop et al. 2001), individual reefs with experimental transplants of corals and other organisms were fertilised for a two year period in the early 1990s. A complex set of effects occurred (e.g., Ward and Harrison 2000; Koop et al. 2001), the meaning of which have been debated for the last 20 years (e.g., Bell et al. 2007).

Overall it is likely that the GBR is eutrophic at certain times and locations, especially when the criteria for eutrophication are based on conditions for coral reef waters (Brodie et al. 2011). Thus if we use criteria such as sufficient nutrient enrichment to cause:

- enhanced crown of thorns starfish outbreaks (Brodie et al. 2005; Fabricius et al. 2010);
- enhanced thermal coral bleaching response (e.g., Wooldridge and Done 2009; Wooldridge 2016) and changed energetic relationships between corals and zooxanthellae (Wooldridge 2013);
- increased bioerosion (Hutchings et al. 2005);
- increased turbidity (in association with increased fine sediment supply) (Fabricius et al. 2014, 2016);
- changed dominance of macroalgae over coral growth (De'ath and Fabricius 2010);
- changed phytoplankton species composition (Furnas et al. 2005; Jones et al. 2016); and
- increased incidence of some coral diseases (Bruno et al. 2003; Haapkyla et al. 2011),

then it is certain that some degree of eutrophication already exists in the GBR, especially during intense river runoff. However, eutrophication in coral reef situations is a complex issue (Fabricius et al. 2013a; D'Angelo and Wiedenmann 2014; Risk 2014) and controversy over the state of eutrophication of the GBR will, no doubt, continue.

3.3 Debates over loads of materials/pollutants delivered to the GBR in river discharge; the sources of these pollutants in catchments; the contributions from different land uses; the contributions from different types of erosion; the suitability of catchment models in use; and the balance between modelling and monitoring

Reliable estimates of the loads of substances such as fine sediment, nitrogen and phosphorus (and their various forms), and pesticides discharged from the rivers flowing into the GBR have been difficult to achieve (Brodie et al. 2012a). Accurate load estimates are required to establish material budgets for the GBR (e.g., Furnas et al. 2011), to assess how loads have changed since catchment development started in the 1830s (Kroon et al. 2012) and hence to evaluate the "anthropogenic load" and thereby set targets for load reduction (e.g., Brodie et al. 2014; Wooldridge et al. 2015).

Equally important are accurate estimates of the sources of the loads by, for example, land use (e.g., Waterhouse et al. 2012), sub-catchment (Bainbridge et al. 2014, 2016; Bartley et al. 2014b; Croke et al. 2015), agricultural management practice and erosion type (e.g., gully, hillslope, streambank, channel, rill) (Waters et al. 2014; Wilkinson et al. 2013, Bartley et al. 2014a, b; Olley et al. 2013; Brooks et al. 2009, 2014, 2015; Piesch et al. 2015). As the Burdekin River has been extensively studied for load estimation, here I use load estimates for fine sediment in that river to show how estimates have changed through time. Belperio (1979) used a regression-based sediment rating-curve to calculate an annual average load of 3.45 million tonnes of wash load (clay and silt) for the Burdekin region, using monitoring data from the 1970s. Subsequently, annual average suspended sediment load estimates for the Burdekin River have been derived using monitoring data (estimates of 3.8 – 4.6 million t yr^{-1}) and catchment models (2.4 – 9.0 million t yr^{-1}) with some models also predicting "natural" loads (0.48 – 2.1 million t yr^{-1}) (Table 3) (Brodie et al. 2009a). Therefore, there has been reasonable agreement on the order of magnitude of the fine sediment load, with only a few outliers, but also a gradual refinement in the estimate and its reliability and some resolution of the arguments about whether "monitoring" or "modelling" give the best estimate of long-term average loads.

Table 3. Annual average suspended sediment load estimates for the Burdekin River, 2009 – 2014, indicating methods, sources and caveats.

Load (millions of tonnes)	Method	Reference	Caveats
3.0	Rating curve	Belperio 1979	Pre Burdekin Falls dam
2.8	Simple catchment model	Moss et al. 1992	Burdekin plus Haughton
9.0	Catchment model	Neil & Yu 1996	Burdekin plus Haughton
3.8	Monitoring with a modelling generalisation step	Furnas 2003	
8.0	Catchment model	Neil et al 2002	Burdekin plus Haughton
2.8	SedNet	Brodie et al 2003	Also: McKergow et al. 2005a,b
2.8	SedNet	Cogle et al 2006	
4.5	Compilation of estimates with quality control	Brodie et al 2009b	
4.0	Compilation of estimates with quality control	Kroon et al 2012	
6.5 – 12.7	Monitoring and linear interpolation	Joo et al 2012	Loads for 3 specific years in the period 2006 – 2009 (high rainfall years)
0.9 - 15	Linear regression estimator	Kuhnert et al 2012	Loads for 23 specific years in the period 1986 - 2010
3.9	Compilation of estimates with quality control	Bainbridge et al. 2014	
4.0	Source Catchments model	Dougall et al 2014	Also: Waters et al 2014

There has also been considerable debate over the sources of sediment, nutrients and pesticides in rivers with the respect to land-use, industry and erosion processes, and the proportion of the total end-of-valley load from each. It is well demonstrated that most of the anthropogenic nitrate loads come from losses of fertilisers from sugarcane cultivation, that much of the anthropogenic fine sediment load comes from erosion in grazing lands and that the main discharge of pesticides (particularly PSII herbicides) results from use in the sugarcane industry (Waterhouse et al. 2012; Waters et al. 2014); however, the contributions of forest, woodland, pasture and introduced legume pasture, urban areas, horticulture, mines, grain cropping, cotton and aquaculture are still vigorously debated particularly by those involved in these industries or land uses.

3.4 Debate over seaward extent of terrestrial material in the GBR

Opinions about the dispersion of material (sediment, nutrients, pesticides, metals, organic chemicals) carried in terrestrial runoff and the extent of dispersal into the GBR lagoon, have differed greatly through time. Wolanski and van Senden (1983) and Wolanski and Jones (1981) showed that river plumes from the Burdekin and Wet Tropics rivers could be detected through their reduced salinity signal for hundreds of kilometres to the north of river mouths and well offshore, but whether sediment and nutrients were transported so far was not examined (Wolanski et al. 1984; Johnson and Carter 1988; Gagan et al. 1987, 1988, 1990). Most terrestrial sediment deposited on the floor of the GBR lagoon forms a band within 15 km of the coast (Belperio 1983). Some studies suggest that terrigenous input reaches only halfway across the shelf while others have found terrigenous marker chemicals extending to the edge of the shelf break (e.g., Curry and Johns 1989). In general, there appears to be an inner reefal area dominated by terrestrial sediment and an outer area dominated by carbonate sediment (Johnson and Carter 1988; Wolanski and van Senden 1983).

Strong statements as to the spatial extent of terrestrial material transport across the GBR shelf towards the shelf-break were made following studies into the effects of Cyclone Winifred (Furnas and Mitchell 1986), a category 3 storm which crossed the GBR near Innisfail in early 1986, producing widespread benthic sediment resuspension (Gagan et al. 1988, 1990) and major river discharges into the GBR lagoon (Gagan et al. 1987), followed by a widespread phytoplankton bloom (Furnas 1989). Gagan et al. (1987) claimed there was very little transport of materials from river discharge (the Johnstone River primarily) across the shelf:

"Tropical Cyclone Winifred (1 February 1986) provided an ideal opportunity to examine the fate of high river discharge in the Central Great Barrier Reef by producing near-record floods between Townsville and Cairns. Comparison of the carbon isotope ratio of organic matter in shelf sediment collected immediately before and after the cyclone showed that the bulk of terrestrial plant detritus from the Johnstone River was deposited within 2 km of the river mouth and none moved more than 15 km offshore. By comparing the magnitude of the Johnstone River flow to the maximum recorded flows of other rivers in the Great Barrier Reef Province, ***we conclude that terrestrial runoff has not reached the Reef in historical times*** *except, perhaps, during rare Burdekin River floods."* [My emphasis added]. In this context "the Reef" is used to mean the mid and outer-shelf reefs of the GBR (and not the inner-shelf reefs). In contradiction of this statement, in the 30 subsequent years it has become clear that terrestrial runoff from large river discharge events reaches the mid and outer shelf of the GBR almost every year, but mostly as dissolved, not particulate, material (e.g., Larcombe et al. 1996; Devlin and Schaffelke 2008; Alvarez-Romero et al. 2013; Devlin et al. 2012a).

One major line of enquiry has been the study of river flood plumes and their behaviour in the GBR lagoon (following the studies of Wolanski mentioned above) using in-situ sampling, remote sensing (satellite and airborne) and modelling (Devlin et al. 2015b). Some studies concentrated on the composition of the plumes (Brodie and Mitchell 1992; Brodie et al. 2010), some on the extent of the plumes (Davies and Hughes 1983; Brodie 1996; Schroeder et al. 2012) and some on both extent and composition (Devlin et al. 2001; Devlin and Brodie 2005; Devlin and Schaffelke 2009; Devlin et al. 2012a), while more recent efforts aimed to characterise water types and their potential effects on GBR organisms (Devlin et al. 2013; Alvarez-Romero et al. 2013). Other studies have used plume data and biological assessments to determine the effects of river discharge on coral reefs (Jones and Berkelmans 2014; Butler et al. 2013, 2015; Wenger et al. 2016) and seagrass meadows (Preen et al. 1995; Petus et al. 2014) and seagrass-associated biota such as dugong (Preen and Marsh 1995). Other studies used coral cores and the chemicals preserved in their annual banding to assess the influence of river discharge (e.g., McCulloch et al. 2003; Lough 2007, 2008; Lewis et al. 2012a).

From these studies it is clear that some materials discharged from rivers easily reach the mid-shelf reefs (e.g., from the Fitzroy River to Heron Island – Brodie and Mitchell 1992) and even the outer-shelf reefs and the Coral Sea occasionally (e.g., to the Coral Sea in 2008 – Devlin and Schaffelke 2009). Satellite images from 2007 clearly show plumes (visible due to algal blooms) dispersing across the shelf through the outer-shelf reefs and well into the Coral Sea (Figure 3) (Brodie et al. 2011, 2012b). As low-salinity water and its content of humic acids spread across the shelf during flood plumes, luminescent lines form in growing corals along with density banding due to seasonal growth (e.g., Lough 2011a) in massive corals such as *Porites* spp. The luminescent lines can be used as a signal of both the magnitude of river influence across the shelf (Lough 2011b) and the humic acid content of the discharge water. Studies of this type have identified river discharge effects right across the continental shelf (Lough et al. 2002). Recent studies have shown that the magnitude of river discharge has increased greatly in the period since about 1850, associated with large-scale oceanographic/climatic features in the Pacific Ocean affecting rainfall intensity (Lough et al. 2015).

Another line of enquiry has examined sediment dispersal and deposition. The early consensus was that most sediment discharged from rivers deposited quite close to the river mouth (e.g., Gagan et al. 1989) but that resuspension during strong wind conditions and transport by the prevailing south to north coastal currents (driven by the SE trade winds) moved the sediment up the coast before it was mostly trapped in northward facing bays (Orpin et al 1999, 2004). More recent studies (Lewis et al. 2006; Lewis et al. 2014) have confirmed that most sediment is deposited near river mouths, but have shown that there is little further dispersal. However, a small fraction of the fine sediment is transported in the flood plumes (Bainbridge et al. 2012) for large distances up the coast, forming rich organic flocs during transport and being largely responsible for the extra turbidity in coastal areas like Cleveland Bay, where they may last for a year or more following major discharges from the Burdekin River (Fabricius et al. 2014, 2016; Logan et al. 2013, 2014).

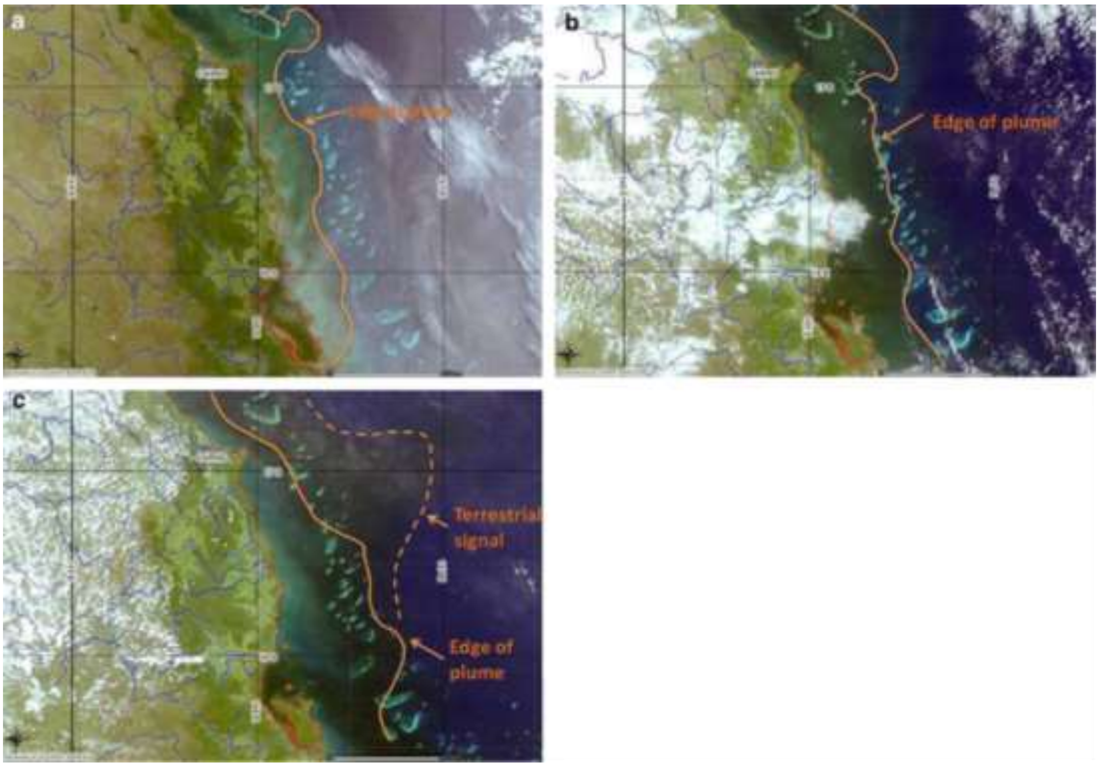

Figure 3. Progression (a–c) of a multiple river plume in the Wet Tropics (9, 11, 13 February 2007, respectively) extending from the coast to beyond the outer reef. The lines show the outer edge of the plume made visible by coloured dissolved organic matter and phytoplankton. Images show the transformation from a plume dominated by terrestrial particulate matter into one dominated by a phytoplankton bloom driven by dissolved nutrients. A proportion of the nutrients in the plume may be seen 'escaping' to the Coral Sea in image c. Image courtesy of CSIRO. From Brodie et al. (2011, 2012b).

3.5 Debate over the relationship between increased nutrient loading and the frequency and severity of crown-of-thorns starfish outbreaks

The crown-of-thorns starfish (*Acanthaster planci* – COTS) is a specialised coral feeder found across the Indo-Pacific. Populations of COTS display cyclic oscillations between extended periods of low density, with individuals sparsely distributed, and episodes of unsustainably high densities, commonly termed "outbreaks". The outbreaks result in mass mortalities of corals with second-order and long-term consequences on various reef communities. COTS outbreaks spread across the reef ecosystem by larval transport, and commonly lead to increases in benthic algal density, loss of coral-feeding assemblages, overall collapse of reef structural complexity, and decline in biodiversity and productivity (Birkeland and Lucas 1990). COTS have caused widespread damage to many coral reefs in the Indo-Pacific over the past six decades as population "explosions" have occurred at regular intervals (Zann et al. 1987, 1990; Birkeland and Lucas 1990; Pratchett et al. 2014). However, our knowledge of the density of COTS on reefs before 1960 is very limited. Given the huge numbers of animals found on reefs across the Indo-Pacific by the 1970s (Birkeland and Lucas 1990), it was postulated that these outbreaks could not have occurred in the past, or not at the frequency at which they were occurring by the 1990s. Hence a range of anthropogenic causes were postulated (Birkeland and Lucas 1990; Brodie 1992).

Outbreaks on the GBR have occurred from 1962 to 1976, 1979 to 1991, 1993 to 2005 (Brodie et al., 2005; Fabricius et al. 2010) and 2009 – present (Pratchett et al 2014; Wooldridge and Brodie 2015). Each outbreak has severely reduced coral cover, especially in the central section of the GBR (De'ath et al. 2012), but the cause (or causes) of the outbreaks remains a controversial issue. One view postulates that population outbreaks are a natural phenomenon due to the inherently unstable population sizes of highly fecund organisms such as COTS (Potts 1981). Conversely, outbreaks are blamed on anthropogenic changes to the environment of the starfish, with a range of possible causes, including: removal of predators on adult starfish (particularly fish and large gastropods (Sweatman 1995; Mendonca et al. 2010); changes to population structure of predators on larval and juvenile COTS, caused by chemical (possibly pesticide) pollution (Chesher 1969; Randall 1972); destruction of predators on larval COTS, particularly corals (which may feed on plankton), by construction activities on reefs; and larval food supply (phytoplankton) enhancement from nutrient-enriched terrestrial runoff (Lucas 1982; Bell and Gabric 1991; Brodie 1992; Brodie et al. 2005; Fabricius et al., 2010).

Birkeland (1981) hypothesised that COTS population outbreaks were more common on high islands in the Pacific due to runoff of terrestrial nitrogen and phosphorus than they were on atoll islands were there is no comparable runoff. The hypothesis was founded on the fact that COTS have a planktonic larval stage (of a few weeks) which feeds on phytoplankton and needs a certain amount of phytoplankton to reach a level of viability that allows settling on a reef. Nutrient runoff in river discharge from high islands can provide the conditions needed for phytoplankton to bloom and provide the required food (Birkeland 1982; Brodie 1992). In "normal" nutrient conditions the larvae have insufficient phytoplankton food to reach competence (Lucas 1982).

Anthropogenic nutrient enrichment is derived from sewage discharge, fertiliser runoff and increased erosion of nutrient-rich soil and leads to enrichment of coastal waters, resulting in higher phytoplankton biomass, a shift to larger species of phytoplankton more suitable as COTS larval food and better survivorship of COTS larvae leading to more frequent outbreaks (Brodie 1992, Brodie et al. 2005). The possibility of this as an explanatory hypothesis for the large outbreaks of COTS on the GBR was first raised in the 1970s (Lucas 1982) but was by then only one of several hypotheses as to the cause of the outbreaks, given that the apparently increased frequency of COTS outbreaks on the GBR and elsewhere in the Indo-Pacific was unlikely to be a natural occurrence (Potts 1981; Moran 1986).

Analysis of larval growth, the effects of environmental factors and experimental testing of the nutrient enrichment hypothesis was first carried out by Lucas and colleagues in the 1970s and showed that COTS larvae were food limited, not developing well in conditions of low nutrient concentrations and phytoplankton biomass (Henderson and Lucas 1971; Lucas 1973, 1982). However, subsequent experiments suggested no nutrient effect (Olson 1987), seemingly challenging the hypothesis, but there were doubts expressed about the Olson results and in the late 1980s the experiments were repeated, this time with more care given to strict protocols regarding nutrient supply. The new results did support the nutrient enrichment hypothesis (Ayuki et al. 1997; Okaji et al. 1997a, b; Fabricius et al 2010). COTS outbreaks associated with anthropogenic nutrient enrichment are now seen as one of the signals of partial eutrophication of the GBR (Brodie et al.

2011; Fabricius 2011). Recent refinements of the experiments have further supported the hypothesis (Uthicke et al. 2015a; Wolfe et al. 2015a). However, it has been shown recently that oceanographic conditions associated with the ENSO cycle (Hock et al. 2014) may play a part in the Cairns area of the GBR, and Wooldridge and Brodie (2015) have confirmed that both nutrient enrichment from river runoff and connectivity due to ENSO conditions are important in initiating outbreaks.

COTS outbreaks can occur at sites with natural nutrient enrichment from ocean upwelling (e.g., in the Chagos archipelago – Roche et al. 2015), at oceanic nutrient/productivity fronts (Houk et al. 2007; Houk and Raubani 2010) or at sites without any apparent nutrient enrichment (Miller et al. 2015), in places where upwelling may be present but not documented. The prolonged COTS outbreak in the Swains reefs in the southern GBR has long been thought to be associated with the upwelling systems in that region (Brodie et al. 2005). Evidence of the biological effects of this upwelling and intrusive activity near the Swains and Capricorn-Bunker group reefs associated with the Capricorn Eddy (Weeks et al. 2010; Mao and Luick 2014) have recently been demonstrated via plankton blooms (Alongi et al. 2015) associated with enrichment phenomena such as manta ray feeding aggregations (Weeks et al. 2015). It is likely that these factors help explain the COTS outbreaks in the Swains and Capricorn-Bunker Group reefs (Miller et al. 2015).

While it has been suggested in the past that seemingly simultaneous periods of COTS outbreaks occur across the entire Indo-Pacific in response to ocean-scale phenomena such as ENSO (L. Zann pers. com.), recent research in Polynesia suggests that COTS outbreaks can have a suite of specific local causes as well as regional-scale connectivity in populations and outbreak phenomena (Timmers et al. 2010, 2012; Kayal et al. 2012). Similarly, in Guam, COTS outbreaks seem to have elements both of a primary outbreak nature, probably with local causes, and of a secondary outbreak nature, with larval recruits possibly coming from places a great distance from Guam (Tusso et al. 2015). The occurrence of several genetically distinct sub-species of COTS in the Indo-Pacific supports the notion of the regional or local nature of outbreaks (e.g., Vogler et al. 2012).

Direct intervention to kill COTS by collection or injection of toxic chemicals has been tried in many places during COTS outbreaks without significant effects on overall population numbers. Between 1970 and 1983, almost 13 million COTS were removed from the reefs of the Ryukyu Islands, southern Japan, via a bounty for fishers, who changed from fishing to a more reliable income (Lucas 2013). Despite this huge effort, there are still large COTS populations in the Ryukyus. However, the idea of direct killing remains attractive to some and has been shown to work at the scale of a tourist site on the GBR (Great Barrier Reef Marine Park Authority 2009a). COTS can be killed by injection of sodium bisulphate, bile salts (Rivera-Posada et al. 2014) or lemon juice and vinegar (Moutardier et al. 2015). However, it is well known that such methods, while effective at keeping COTS away from a tourist reef site, cannot control populations at the population (GBR) scale. The sheer numbers of COTS larvae, their widespread distribution and their adaptability to environmental conditions also makes the case that the success of direct killing as a population level control is unlikely (Doherty et al. 2015; MacNeil et al. 2016; Uthicke et al. 2015b; Wolfe et al. 2015b) despite claims to the contrary from the proponents of the direct killing solution in the popular press. Successful control programs have only been achieved where there was a small discrete population, which was tackled quickly with only small numbers removed – example, 225 animals in the Bos et al. (2013) case study on an isolated reef in the Philippines.

COTS outbreaks continue to be a threat to the GBR. There is some evidence that the increase in the area of no-take zones in 2004 has had significant success as COTS numbers on closed reefs are lower than on reefs open to fishing (Sweatman, 2008; McCook et al., 2010; Vanhatalo et al. 2016) (possibly due to the presence of fish predators on juvenile stages). The debate over whether more frequent COTS outbreaks in the central GBR are largely caused by nutrient runoff from agriculture in the Wet Tropics and Burdekin Regions (Wooldridge et al. 2015) is now reasonably settled (Wooldridge and Brodie 2015) but some doubts remain (Pratchett et al. 2014) and the loss of the predators on various life stages of COTS has not been ruled out as a contributing cause.

3.6 Debate over the importance of pesticides as a threat

During the 1980s various sampling programs for pesticide residues, primarily organochlorine insecticides (OCs), were undertaken in the GBR region. Residues were examined in GBR and Coral Sea waters during 1981 (Tanabe et al., 1982) with relatively low concentrations of hexachlorocyclohexanes (HCHs) and DDT and its breakdown products found. Lindane (c-HCH) was detected in sediments from the mouth of the Burdekin River in 1984/85 but organochlorine pesticides were not detected in sediment samples collected in Bowling Green Bay or at Lizard Island (Dyall and Johns, 1985). The authors concluded that sedimentary accumulation of organochlorines was confined to the close proximity of coastal sugarcane growing areas.

OCs were detected at low concentrations in:

- COTS from Slasher's Reef and the Bunker Group in the southern section of the Marine Park in 1970 and 1971 (McCloskey and Duebert, 1972);
- Dugongs, with very low concentrations of lindane and dieldrin in the livers of four animals collected from Townsville in 1977 (Heinsohn and Marsh, 1978); and
- various reef animals, with low concentrations of c-lindane, heptachlor and DDT present in hard corals (*Fungia* sp. and *Acropora* sp.), liver and muscle tissue from coral trout (*Plectropoma maculatum*), surf parrotfish (*Scarus fasciatus*) and a bivalve mollusc (*Tridacna crocea*) collected from reefs between Heron and Lizard Islands in 1976 and 1977 (Olafson, 1978).

OC insecticides were detected in the Burdekin River and groundwater of the lower Burdekin floodplain in the 1970s (Brodie et al. 1984). Concern about OC pollution of the GBR resulted in a workshop in 1985 (focussed on a range of contaminants including OCs), which noted that the available data was insufficient to cause strong concern but that the situation needed continuing monitoring and assessment (Dutton 1985).

In 1991, after I joined GBRMPA with a "mandate" to look more closely at all water quality issues in the GBR, I made an assessment (with advice from colleagues) that OC insecticides were not a significant threat to GBR ecosystems. In addition, I determined that the pesticides in common use in 1991 (mainly herbicides and organophosphate insecticides in the sugarcane industry) were unlikely to be a threat to GBR ecosystems as their short half-lives would preclude them from being transported in significant amounts to the marine environment. In hindsight, while the assessment with respect to the OC insecticides was accurate, the assessment for herbicides was not. It took another seven years and, fortunately, the appointment of David Haynes to the Water Quality Group in GBRMPA for this assessment to be corrected, with a series of studies carried out by Haynes and his colleagues from ENTOX, led by Jochen Mueller, in the late 1990s (e.g., Haynes et al. 2000a). Our knowledge of pesticides in the GBR was summarised in a workshop in 2000 (Haynes and Michalek-Wagner 2000) and in the keynote paper of Haynes and Johnson (2000). The workshop provided the foundations for the pesticide risk assessment studies in the period from 2000 – 2015 (Devlin et al. 2015a).

Kannan et al. (1995) showed that concentrations of chlorinated organics (PCBs, DDTs, HCHs, aldrin, dieldrin and chlordanes) in muscle tissue of coastal fish species collected near Townsville between 1989 and 1993 were low compared to samples from the Brisbane region and other urbanised centres. Further analysis was carried out in 1992 and 1993 of fish livers from 142 individuals of a wide range of species collected in the central section of the GBRMP (Von Westernhagen and Klumpp, 1995). Low levels of DDE and dieldrin were detected in 8% of samples.

Lindane, dieldrin and DDT (and its breakdown product DDE) continue to be found in nearshore marine samples collected along the Queensland coast (Haynes et al. 2000a). For example, dieldrin was detected in sediments collected from the mouths of the Barron and Johnstone Rivers and in sediments from Halifax Bay. Dieldrin was a widely distributed contaminant of Queensland waterways and estuaries in the past (Clegg 1974; Kannan et al. 1995; Russell et al. 1996) and was detected in crabs (*Scylla serrata*) collected from estuaries adjacent to agricultural catchments between Moreton Bay and Cairns (Mortimer 2000; Negri et al. 2009) and in fish livers collected from the central Queensland coast adjacent to agricultural activity (von Westerhagen and Klumpp 1995).

More revealing than the studies on OCs, however, were the studies of Haynes, Mueller and colleagues over the next 15 years, which showed widespread contamination with photosystem II (PSII) herbicides of GBR sediments and seagrass (Haynes et al. 2000a) and fresh, estuarine and marine waters (e.g., McMahon et al. 2005; Shaw et al. 2010; Kennedy et al. 2012b; Kennedy et al. 2012a; Smith et al. 2012). It was shown that during high river flows, tonnes of herbicides such as diuron and tebuthiuron could be discharged into the GBR lagoon (Mitchell et al. 2005; Packett et al. 2009). Herbicides (PSII and others) were found in highest concentrations in fresh and estuarine waters immediately downstream from large cropping districts (mostly sugarcane cultivation). A series of studies in the lower Burdekin cane growing district (where some other crops are grown as well) showed above-guideline concentrations of many pesticides (Davis et al, 2008, 2012, 2013, 2014a; Smith et al. 2012; O'Brien et al 2016) and high risks of ecosystem damage in some cases (Davis et al. 2013, 2014b). Studies over the last 4 years have confirmed similar contamination in the lower Herbert River floodplain (O'Brien et al. 2013, 2014, 2015).

Controversy over the impacts of herbicide pollution on mangroves arose following studies showing elevated diuron concentrations were responsible for mangroves near Mackay showing a "die-back" condition (Bell and Duke 2005; Duke et al. 2005). There was no question that diuron was present in large quantities in runoff events from the Pioneer River in Mackay (Mitchell et al. 2005). However, conflict arose as to whether the likely cause of the die-back was diuron (and other herbicides) or some other cause such as flooding and burial (Kirkwood and Dowling 2002; Dowling 2008). McKillup (2008) raised concerns about the statistics in the Duke et al. (2005) study; Duke (2008) responded with appropriate corrections. However, questions remained about the causes of the die-back given that the mangroves seemed to be recovering (Abbot and McKillup 2010) although diuron pollution continued. Eventually Abbot and Marohasy (2011) claimed that "*Evidence from field studies suggests* *burial of pneumatophores, the plant's breathing roots, following flood events is a more likely causal factor in mangrove dieback, whereas any contribution from Diuron remains unproven."* This debate has never been fully resolved.

The debate over the relative threat of pesticide pollution extended to the adequacy of management of pesticides in the GBR region and, by extension, within Australia generally. Currently, management takes place through the Federal regulatory authority, the Australian Pesticide and Veterinary Medicine Authority (e.g., APVMA 2012), with on-ground management a state responsibility. King et al. (2013) claim that the system is very unsatisfactory:

> "*The ad hoc, case-by-case and very slow chemical review process administered by Australia's national pesticide regulator has not effectively assessed or addressed chemical risks to the GBR. Some failures of the current system would be addressed by a systematic re-registration program of the kind in place in the European Union and United States. We conclude that to adequately protect the GBR, given its marine protected area and World Heritage status, both the special management provisions for the area already existing plus an effective national pesticide regulatory regime of the standard of the European Union are the minimum requirements."*

Holmes (2014) notes the total inadequacy of the models used by APVMA to assess and hence regulate the use of diuron in the GBR region:

> "*The environmental risk assessment process used by the APVMA utilised a runoff risk model developed and validated under European farming conditions. However, the farming conditions in the sugarcane regions of the Great Barrier Reef catchments have environmental parameters beyond the currently validated bounds of the model. The use of the model to assess environmental risk in these regions is therefore highly inappropriate, demonstrating the pitfalls of a one size fits all approach."*

Camenzuli et al. (2012) showed that a model which took into account the conditions in north Queensland (in the Tully catchment) was able to be parametised and gave diuron results in agreement with monitored data. However, even though pesticide residues far in exceedance of ANZECC guidelines continue to be found in GBR waterways (Devlin et al. 2015a), regulation through APVMA continues to be ineffective. Fortunately, other ways to manage pesticides in the GBRCA using the power of Reef Plan (Department of the Premier and Cabinet 2013) have been implemented and, as a result, pesticide loads discharged to the GBR have declined by 30.5% over the five-year period (2009 – 2014) of Reef Plan 2009 (Department of the Premier and Cabinet 2014), although the Reef Plan 2009 target of 60% has not been achieved. Effective ways to reduce herbicide loss from cane farms have now been tested (Oliver et al. 2014; Davis and Pradolin 2016; Melland et al. 2015). Using banded spraying, in which the diuron is only sprayed on the mound of the sugarcane and other herbicides are sprayed in the inter-row, quantities of diuron and atrazine applied can be greatly reduced, reducing herbicide losses by 90%, without affecting cane productivity (**Oliver et al 2014**). However, banded spraying may have unforeseen consequences when the alternative herbicides (to diuron) are found to be equally or more toxic than diuron (Davis et al. 2014a) and hence present an equal or greater risk to aquatic ecosystems.

The significance of pesticide pollution to the GBR has been analysed by Stephen Lewis and colleagues (Lewis et al. 2009) and the cumulative risk of multiple pesticides analysed (Lewis et al.

2012b; Davis et al. 2013). These analyses and other studies have shown that due to the frequency and duration of exceedance of water quality guidelines, pesticides, particularly herbicides, although continually present in ngL^{-1} concentrations throughout the GBR waters (Shaw et al 2012; Kennedy et al. 2012b; Kennedy et al. 2012a; Smith et al. 2012; O'Brien et al. 2016) are primarily a threat to fresh, estuarine and coastal waters, seagrass meadows and inner-shelf reefs, and only a minor threat to mid-shelf and outer-shelf reef systems (Devlin et al. 2015a).

The insecticide imidacloprid was introduced into sugarcane cultivation in the GBR catchment in the last decade to replace chlorpyrifos as the principal control of cane beetle larvae (Allsopp 2010). Imidacloprid is very effective against these larvae (Chandler 2003) and the Australian sugarcane industry relies on this insecticide in many areas (Hunt et al. 2012). Unfortunately, imidacloprid is notorious in Europe and elsewhere because of its toxicity to bees, jeopardising both crop pollination services and the honey industry (Dively et al. 2015), and in 2013 the EU restricted its use. Its effects on aquatic fauna in the GBRCA and GBR are currently unknown.

4. Discussion regarding water quality guidelines for GBR waters and suitable pollutant load targets for GBR rivers and the GBR generally

4.1 Water quality guidelines

Water quality guidelines were set for Australian waters in 1974 (Hart 1974). From this initial work the Australian and New Zealand Environment and Conservation Council developed "official" guidelines for Australian and New Zealand waters in 1992 and again in 2000 (ANZECC and ARMCANZ 2000), including guidelines for tropical marine waters but not specifically for the GBR. GBRMPA commissioned studies to develop guidelines specifically for GBR waters, which were based on an evaluation of the tolerance of corals to nutrients and related water quality variables (Greenfield et al. 1987; Hawker and Connell 1989; Bell et al 1989). However, no official guidelines were published at this time.

In 2005 Moss et al. (2005) produced a new set of draft guidelines for GBR marine waters, incorporating some guidelines from the ANZECC document but also devising new ones using current knowledge of GBR marine water quality. For example, a guideline of 0.6 μgL^{-1} was set for chlorophyll a, based on current knowledge of chlorophyll a from chlorophyll monitoring programs (Brodie et al. 1997, 2007) in less developed regions of the GBR, where chlorophyll concentrations were thought to be near natural. Subsequently a more structured program was established to set water quality guidelines resulting in the report by De'ath and Fabricius (2008). Using the large data sets available on water quality parameters in the GBR lagoon in both river discharge periods (Devlin et al. 2001) and non-discharge periods (De'ath and Fabricius 2008) a team of managers and scientists published the first official GBR water quality guidelines (Great Barrier Reef Marine Park Authority 2010), with the guidelines for chlorophyll a being 0.45 μgL^{-1} (annual) and 0.63 μgL^{-1} (wet season).

Currently, pesticide guidelines for marine waters of the GBR are being established or revised. New guidelines are badly needed as most of the most commonly used pesticides used in cropping and grazing in the GBRCA have no guidelines, and for those few that do, the guidelines are unsuitable. Australia's pesticide regulatory system is badly deficient in ensuring pesticide residues are of minimal danger to aquatic environments (King et al. 2013; Holmes 2014). The Queensland

Government is now working on guidelines for a range of common pesticides (mostly herbicides), which should be incorporated into ANZECC and GBRMPA guidelines (R. Smith pers. com.).

4.2 Pollutant load reduction targets

4.2.1 Target setting in the period 2001 - 2015

The first targets for load reductions of pollutants discharged from rivers to the GBR were made in the Great Barrier Reef Water Quality Action Plan (Brodie et al. 2001a). Reduction targets were based on the assessed degree of anthropogenic modification of sediment, nitrogen and phosphorus inputs to individual GBR catchments. For catchments having more intensive development and hence a larger increase in delivery compared to pre-development times (circa 1850), higher targets were set – for example, 50% reductions in total load of suspended sediment. Catchments with little development were required to reduce loads by only small percentages. The loads for each river were based on use of the SedNet model as used in the National Land and Water Resources Audit (NLWRA 2001). For pragmatic and political reasons, Reef Plan 2003 did not contain numerical load reduction targets and at that stage there were no GBR-specific marine water quality guidelines either. The ANZECC 2000 guidelines (ANZECC & ARMCANZ, 2000) were in place by 2003 and included marine water quality guidelines for nutrients and metals but were not designed to be applied in tropical waters generally or, specifically, in GBR waters.

It was not until the first revision and update of Reef Plan in 2009 following the Scientific Consensus Statement of 2008 (Brodie et al. 2008a) that the first load targets and land management targets were set under Reef Plan. End-of-system load targets for the major pollutants were set for the entire GBR in Reef Plan 2009 (Department of the Premier and Cabinet 2009) and load targets were set within the Reef Rescue initiative in 2008 (Brodie et al. 2012a) (Table 4). The two sets of targets are loosely linked although internally inconsistent. Both sets of targets were based on what could be achieved through “feasible” agricultural management change to “better” management practices of the Great Barrier Reef Catchment (GBRC) (Brodie et al. 2012a). End-of-system load targets for the major pollutants addressed in Reef Plan 2009 were updated in 2013 (Department of the Premier and Cabinet 2013). Targets were not established on the basis of ecological realities for the GBR although attempts to design targets of this type have been made (e.g., **Brodie et al. 2009a**). There is no guarantee that the Reef Plan 2009 or Reef Plan 2013 targets will lead to the overall Reef Plan objective of “To ensure that by 2020 the quality of water entering the reef from adjacent catchments has no detrimental impact on the health and resilience of the Great Barrier Reef”. Reef Plan 2013 includes water quality targets and land and catchment management targets to be achieved by 2018 (Table 5).

Targets at a basin scale were not set during Reef Plan 2009 or Reef Plan 2013. Thus there are no formal Reef Plan targets for the individual basins of the GBR catchment.

The possibility of achieving the overall goal of Reef Plan of "no detrimental impact" is also in question given that current "Best Management Practices" may be insufficient (Kroon, 2012; Thorburn and Wilkinson, 2013). Modeling of land-use adoption scenarios across the entire GBR has shown that complete adoption of current best management practices in grazing and sugarcane would be sufficient to meet the Reef Plan targets for photosystem II herbicides, but the effects are uncertain for suspended sediment, nitrogen and phosphorus (Thorburn and Wilkinson 2013; Thorburn et al., 2013a; Waters et al. 2013) and for the desired ecological outcomes (Kroon 2012).

Table 4. Reef Plan (2009) and Reef Rescue (2009) targets. Reef Rescue targets are shaded. EOC = end of catchment.

Target	Scale (area) for reporting	Reporting frequency
50% Reduction in N load at EOC by 2013	EOC for all GBR catchments	Annual
50% Reduction in P load at EOC by 2013	EOC for all GBR catchments	Annual
Reduce the load of dissolved nutrients from agricultural lands to the GBR lagoon by 25% by 2013	EOC for all GBR catchments	Annual
Reduce the discharge of particulate nutrients from agricultural lands to the GBR lagoon by 10% by 2013	EOC for all GBR catchments	Annual
50% Reduction in pesticide load at EOC by 2013	EOC for all GBR catchments	Annual
Reduce the load of chemicals from agricultural lands to the GBR lagoon by 25% by 2013	EOC for all GBR catchments	Annual
Minimum 50% late dry season groundcover in dry tropics grazing lands by 2013	Sub catchment for Burdekin and Fitzroy	Annual
20% Reduction in sediment load by 2020	EOC for all GBR catchments	Annual
Reduce the discharge of sediment from agricultural lands to the GBR lagoon by 10% by 2013	EOC for all GBR catchments	Annual
No net loss or degradation of wetlands	Catchment for all GBR catchments	Yr 1 and 5
Condition and extent of riparian areas improved	Catchment for all GBR catchments	Yr 1 and 5
80% of landholders adopted improved practices	Sub catchment or catchment by sector	Annual
To increase the number of farmers who have adopted land management practices that will improve the quality of water reaching the reef lagoon by a further 1300 over 3 years	Sub catchment or catchment by sector	Annual progress; major report 2011
50% of landholders adopted improved practices (grazing)	Sub catchment or catchment	Annual
To increase the number of pastoralists who have improved ground cover monitoring and management in areas where runoff from grazing is contributing significantly to sediment loads and a decline in the quality of water reaching the reef lagoon by a further 1500 over 3 years	Sub catchment or catchment	Annual progress; major report 2011

Table 5. Reef Plan 2013 targets.

Long term goal	To ensure that by 2020 the quality of water entering the reef from broadscale land use has no detrimental impact on the health and resilience of the Great Barrier Reef.
Water quality targets (2018)	At least a 50 per cent reduction in anthropogenic end-of-catchment dissolved inorganic nitrogen loads in priority areas. At least a 20 per cent reduction in anthropogenic end-of-catchment loads of sediment and particulate nutrients in priority areas. At least a 60 per cent reduction in end-of-catchment pesticide loads in priority areas.
Land and catchment management targets (2018)	90 per cent of sugarcane, horticulture, cropping and grazing lands are managed using best management practice systems (soil, nutrient and pesticides) in priority areas. Minimum 70 per cent late dry season groundcover on grazing lands. The extent of riparian vegetation is increased.

In March 2015 the Reef 2050 Long Term Sustainability Plan (LTSP) (Commonwealth of Australia 2015) was released (further discussed in Chapter 10). The LTSP is a joint initiative between the Australian and Queensland Governments and provides an overarching strategy for management of the GBR. It contains objectives, targets and actions across several themes, including: biodiversity, ecosystem health, heritage, water quality, community benefits and governance. The LTSP builds on the Reef Plan 2013 targets as indicated below, with the extended LTSP targets in boldface:

- at least a 50 per cent reduction in anthropogenic end-of-catchment *dissolved inorganic nitrogen* loads in priority areas, **on the way to achieving up to an 80 per cent reduction in *nitrogen* in priority areas by 2025**;
- at least a 20 per cent reduction in anthropogenic end-of-catchment loads of sediment in priority areas, on the way to achieving up to a 50 per cent reduction in priority areas by 2025;
- at least a 20 per cent reduction in anthropogenic end-of-catchment loads of particulate nutrients in priority areas; and
- at least a 60 per cent reduction in end-of-catchment pesticide loads in priority areas.

In addition, the Queensland Government announced an election commitment in 2015 that adopted and extended these targets:

- Reduce nitrogen run-off by up to 80% in key catchments such as the Wet Tropics and the Burdekin by 2025; and
- Reduce total suspended sediment run-off by up to 50% in key catchments such as the Wet Tropics and the Burdekin by 2025.

While the Reef Plan targets refer to reductions in "anthropogenic end-of-catchment" loads, and define the pollutants as "dissolved inorganic nitrogen" and "sediment and particulate nutrients", the

LTSP long-term targets and the current Queensland Government targets are less specific, using the term “up to” and referring only to “nitrogen” and “sediment” and thus lend themselves to mixed interpretations. Both sets of targets refer to “priority areas” or “key catchments”, which also require further definition.

4.2.2 Changes in the targets between 2009 and 2013

Large changes in many of the targets were made in Reef Plan 2013. Targets were greatly relaxed for nitrogen and phosphorus, remained unchanged for sediment and were marginally tightened for pesticides (from 50% to 60% reduction in loads). However, interpreting how the new targets compared to the 2009 targets was difficult as the new targets were often given in different forms of the pollutants – for example, “dissolved inorganic nitrogen” (2013) compared to “nitrogen” (2009). Best estimates of the changed nitrogen and phosphorus targets (Brodie et al. 2014) involved comparing the percentages of PN to DIN and to DON in the Source Catchment estimates for the whole GBR discharge (Waters et al. 2014) and assumed that a reasonable interpretation of the Reef Plan 2013 target of:

> “At least a 50 per cent reduction in anthropogenic end-of-catchment dissolved inorganic nitrogen loads in priority areas”
>
> and
>
> “At least a 20 per cent reduction in anthropogenic end-of-catchment loads of sediment and particulate nutrients in priority areas”

meant a 50% reduction in DIN, a 20% reduction in PN and no reduction in DON, with ‘priority areas’ ignored (as they are not clearly defined in Reef Plan 2013) and replaced by ‘all the GBR catchment (GBRC)’.

Similarly, for phosphorus the Reef Plan 2013 targets of:

> “At least a 20 per cent reduction in anthropogenic end-of-catchment loads of sediment and particulate nutrients in priority areas”

are interpreted to mean no reduction in DIP, a 20% reduction in PP and no reduction in DOP across the GBRC.

Thus, in 2009, a 50% reduction in TN load was required whereas in 2013 only a 36% reduction was required. Similarly, for TP, in 2009 a 50% reduction was required whereas in 2013 only a 16% reduction was required. These changes will have consequences in reporting progress towards targets, as the targets are now much less stringent.

4.2.3 Confusion over targets hinders management

The changes in the Reef Plan targets between 2009 and 2013, especially the changes from one form of “nitrogen” (possibly meaning anthropogenic nitrogen, but not made clear) in 2009 to “anthropogenic dissolved inorganic nitrogen” and “particulate nitrogen” (assumed from the descriptor “particulate nutrients”) have proven an issue for interpreting progress towards the targets. This is clearly seen in the 2014 Reef Report Card (Department of the Premier and Cabinet

2014), which reports progress over the period 2008 – 2013, during which load reductions were achieved with the first tranche of Reef Plan funding, but reporting is done against the 2013 targets (which are really targets for 2018) rather than the 2009 targets, which was the original intent of the reporting process. For example, the Reef Plan 2009 target of " a 50% Reduction in N load at EOC by 2013" (Table 4), is reported against in the 2014 Reef report Card (Department of the Premier and Cabinet 2014) as a 17% reduction in DIN and a 11.5% reduction in PN. Targets for DIN and PN are set for 2018 in Reef Plan 2013 but no targets for these nitrogen forms were set in Reef Plan 2009.

This confusion over the wording and meaning of the Reef Plan (and LTSP) targets has led to a process to set "ecologically relevant targets" for Water Quality Improvement Plan (WQIP) processes as this type of target is ecologically based and clear in definition of the relevant parameters, and can be clearly described (Brodie et al. 2016), as shown below.

4.2.4 Setting Ecologically Relevant Targets

Ecologically relevant targets (ERTs) attempt to define the catchment-specific pollutant load reductions that would be required to meet the standards of the GBR Water Quality Guidelines (GBRMPA 2010), which are considered to be suitable to maintain ecosystem health. Thus ERTs are required to be met to achieve the overall long-term Reef Plan goal of "To ensure that by 2020 the quality of water entering the reef from broadscale landuse has no detrimental impact on the health and resilience of the Great Barrier Reef". The Reef Plan 2013 Targets (RPTs) are not set with a clear link to achieve this overall goal and are not based on an ecological endpoint for GBR ecosystems (or proxies like water quality).

ERTs were first set for DIN loads from the Wet Tropics rivers in 2006 (Wooldridge et al. 2006) and the results were incorporated into the draft Tully WQIP (Kroon 2009). Unfortunately, as the WQIP was never accepted for implementation, the use of the ERTs in this case was never tested. Methodologies for setting ERTs for other parameters and other rivers were described by Brodie et al. (2009a) although the term "GBR ecosystem targets" was used at the time to describe what we now call ERTs. These methodologies were used in some of the WQIPs of the period 2006 – 2008. In 2013 a decision was made to use ERTs in developing the new set of WQIPs, initially for the Burnett Mary and Wet Tropics Regions and later for the Burdekin and Fitzroy Regions. The methodology chosen followed Wooldridge et al. (2006) and Brodie et al. (2009) and, more recently, Wooldridge et al. (2015) and Brodie et al. (2016).

Ecologically relevant targets have now been widely used in planning for the management of GBR catchments in Water Quality Improvement Plans (WQIPs). ERTs were set for fine sediment, nutrients and pesticides in the Burnett Mary WQIP (**Brodie and Lewis 2014**; Burnett Mary NRM Group 2015) and the results were used in a benefit-cost analysis of management actions for the Burnett Mary Region (Beverly et al. 2011). ERTs for the Wet Tropics Region were used similarly (Terrain NRM 2015; Brodie et al. 2014), as will be ERTS currently being developed for the Burdekin and Fitzroy regions (Brodie et al. 2015c, 2016; NQ Dry Tropics, 2016).

With this type of target the load reductions needed to achieve a GBR endpoint (and restored ecosystem value) can be analysed in terms of costs of management and benefit-cost ratios (e.g., Beverly et al. 2011).

5 Debate over the influence of new sediment delivery on the turbidity due to benthic sediment resuspension in shallow GBR waters and associated damage to coral reefs and seagrass meadows

5.1 Dynamics of suspended sediments and turbidity

The GBR exists in a sedimentary setting with the shallow (0 – 20m) inner shelf dominated by silico-clastic sediments and the deeper (20 – 100m) mid and outer shelf dominated by carbonate sediments (Hopley et al. 2007). Turbidity on the inner shelf is driven by short-lived flood plumes in the wet season (Devlin et al. 2012a) (Figure 4) and, more importantly, by resuspension of benthic fine sediments and organic particulate material throughout the year, driven by the SE trade winds and tides (Larcombe et al. 1995; Fabricius et al. 2013b, 2014, 2016) (Figure 5). However, the extent of increased turbidity on the GBR inner-shelf caused by increased sediment loads from rivers (associated with modern catchment development) has been in dispute since the early 1990s (Larcombe et al. 1995; Larcombe and Woolfe 1999a,b; Brodie 1996). It was postulated by a group of sedimentologists and physicists that turbidity on the inner-shelf on the "sediment wedge" was not supply limited, there being always adequate fine sediment available for resuspension during SE trade winds and strong tidal currents (Larcombe et al. 1995; Larcombe and Woolfe 1999a,b; Orpin and Ridd 2012). Hence additional sediment loads from rivers would make no difference to the turbidity regime in GBR coastal waters and have no impact on coastal ecosystems. Conversely, a group of biologists, oceanographers and other scientists claimed that river discharge introduced new, finer, "more easily resuspendable" material into the lagoon and that this was the source of the material producing increased turbidity in coastal waters, with consequent effects on light-dependent

Figure 4. Flood plume turbidity: MODIS-Aqua Image of the Burdekin Region during a moderate flood discharge event (10th February 2007). Image provided by NASA and processed by Matt Slivkoff. (From Logan et al. 2013).

Figure 5. Resuspension turbidity: MODIS-Aqua Quasi-True Colour Image of the Burdekin Region, 23 October 2008. (From Logan et al. 2013).

organisms (e.g., Brodie 1996, Fabricius and De'ath 2001a,b; Wolanski and Spagnol 2000; Wolanski et al. 2004). Conclusive evidence for this hypothesis was not produced until long-term studies on coastal photic depth (Weeks et al. 2012) and river sediment discharge (e.g., Kuhnert et al. 2012) were brought together to show the relationship between river fine sediment discharge and annual turbidity regimes (Fabricius et al. 2013b**;** Fabricius et al. 2014, 2016).

Understanding the impacts of sedimentation and turbidity on coral communities and seagrass meadows, and the relationships between end-of-catchment loads and turbidity in the receiving environment is critical to resolving such debates. The suspended sediment of most risk to the GBR (as it is transported furthest in flood plumes, stays in suspension longest (Storlazzi et al. 2015) and results in the greatest degree of resuspension) is the fine fraction, sometimes defined as that smaller than 15.6 µm – that is, the component containing the clay and fine silt fractions (Bainbridge et al., 2012, 2014, 2016; Bartley et al., 2014a; Douglas et al. 2008; Waterhouse et al., 2013). Of even more risk is the clay fraction (<4 µm), which carries most of the nitrogen, phosphorus and other contaminants, travels widely in flood plumes rather than depositing near the river mouth (Lewis et al., 2014, 2015a, b; Delandmeter et al. 2015), is most effective at attenuating light when in suspension (Storlazzi et al. 2015) and drives increased turbidity on the inner and mid shelf of the GBR (Fabricius et al. 2013a, 2014, 2016; Logan et al. 2014).

This increased fine sediment supply can have severe impacts on GBR organisms such as: reef fish, through effects on recruitment and feeding (e.g., Wenger et al. 2011, 2012, 2013, 2014; Hess et al. 2015; Gordon et al. 2015); corals, through sedimentation (e.g., Fabricius and Wolanski 2000; Weber et al., 2006, 2012; Flores et al., 2012; Pollock et al. 2014) and decreased light (Fabricius et al. 2013, 2014, 2016); macro-algae and turf algae through increasing competitive advantage of over corals (Gowan et al. 2014; Goatley and Bellwood 2012, 2013); and seagrass (Collier et al., 2012a,b, Petus et al., 2014). Furthermore, suspended sediment interacts with other stressors to increase the overall impact on coral reefs (Ban et al. 2014; Risk 2014; Graham et al. 2015), and resuspension of sediment by windy conditions or strong tidal currents in shallow waters (<15 m) can bring suspended sediment concentrations above the GBR water quality guidelines (De'ath and Fabricius, 2008; Great Barrier Reef Marine Park Authority, 2010), threatening corals and seagrasses through reduced light for photosynthesis (Bartley et al. 2014).

Some mineral type are particularly important in driving adverse effects offshore (Bainbridge et al. 2016); for example, the expandable clays like smectite are very mobile in suspension in the marine environment (Smith et al. 2008) and form the organically rich flocs (Bainbridge et al. 2012) that are most responsible for far-field resuspension, causing loss of clarity (Fabricius et al. 2014, 2016; Logan et al. 2014) and adverse effects on corals when deposited on to the coral surface (Weber et al. 2006, 2014).

5.1.1 Sediment and corals

Some coral species can tolerate very high sedimentation rates and turbidity, and can recover from short-term or low levels of sedimentation (e.g., Bartley et al. 2014a; Schaffelke et al. 2013; Brodie et al. 2013b). However, most corals are reliant on autotrophy through their associated zooxanthellae and are negatively affected by smothering (sedimentation) and reduced light availability for photosynthesis due to turbidity. Turbidity (and poor water quality generally) can also increase susceptibility to ocean acidification (Uthicke et al 2014; Vogel et al. 2015).

For coral reef systems, reduced water clarity has been associated with increased macroalgal cover, reductions in coral biodiversity (De'ath and Fabricius, 2010), increased macro-bioeroder densities (LeGrand and Fabricius, 2011), shifts from communities dominated by phototrophic corals to heterotrophic filter feeders (Birkeland, 1988), reduced resilience against ocean acidification (Vogel et al. 2015) and increased presence of heterotrophic soft corals compared to autotrophic types as well as loss of soft corals in more turbid waters (Fabricius and De’ath 2001a) and loss of crustose coralline algae (Fabricius and De’ath 2001b). However, some coral reefs have developed and thrived in shallow nearshore areas with high turbidity (Browne et al., 2012; Palmer et al., 2010).

It would appear that coastal reefs like Paluma Shoals and Bramston Reef are sediment-tolerant, having existed and developed in a high-sediment environment since their development during the Holocene (Ryan et al. 2016). Reefs a little further offshore such as Middle Reef and Pandora Reef are also relatively sediment-tolerant, sufficient to be able to survive and recovery from acute damage in a high (and increasing) sediment regime (Browne et al. 2010, 2012; Done et al. 2007), although with periods of low coral cover. This is evident from the state of coral cover on Pandora Reef (Coppo and Brodie 2015 with data from the LTMP (AIMS 2016) and MMP coral monitoring programs (e.g., Thompson et al. 2015)) with generally good (40 - 60%) cover at LTMP sites (1995 – 2012) but low (<10%) cover at MMP sites (2005 – 2014) (Figures 6 and 7) and a history of large variations in cover at different sites on the reef (Done et al. 2007). On Middle Reef coral cover has been maintained in good condition (50%) and fairly stable over 2005 – 2014 (Figure 7) (Coppo and Brodie 2015; Browne et al. 2010).

Further offshore again, at the inner-shelf fringing reefs in the Keppels, Whitsundays, Magnetic, Palms, Dunk islands and elsewhere, reefs may be more sensitive to an increased sediment regime, with less tolerant and adaptable coral species. At such sites coral cover varies from very poor to poor but variable – for example, at Havannah Island (Figure 6) coral cover at LTMP sites has declined from 45% to less than 5% between 1996 and 2014, but at the MMP sites (Figure 7) has increased from 15% to 30% between 2005 and 2014. The reefs in the Keppel Islands group were subjected to almost annual large discharge events from the Fitzroy River, with high concentrations of suspended sediment, nutrients and pesticides over the period 2008 - 2013 (Devlin et al. 2012b; Wenger et al. 2016). As a result, the reefs lost resilience and coral cover has declined to very low levels (Jones and Berkelmans 2014; Wenger et al. 2016).

5.1.2 Sediment and seagrass

Reduced water clarity may cause loss of seagrasses from deeper waters (Collier et al., 2012a,b) or, when the reduced clarity is prolonged, to seagrass mortality (Petus et al. 2014). Seagrass health and abundance is variable in space and time in the GBR (Coles et al. 2015). Recent evidence indicates that seagrass is declining in parts of the GBR (McKenzie et al 2015; Coles et al. 2015), particularly in the Townsville region (McKenzie et al., 2010; Petus et al. 2014), Cairns region (Rasheed and Unsworth 2011; McKenna et al. 2015), Abbot Point region (Rasheed et al. 2014), and several other regions, associated with a series of severe cyclones and large river flood events (McKenzie et al. 2015; Coles et al. 2015). Evidence of this decline is that 38% of sampling sites monitored regularly across the GBR are exhibiting shrinking meadow area, a large number of sites have reduced seagrass abundance, and many sites have limited or no sexual reproduction and so are not producing seeds that would enable rapid recovery (McKenzie et al. 2015). Degraded light regimes from increased suspended sediment are the cause of reduced seagrass abundance in many sites. The 2011 major river discharge events from many of the GBR rivers associated with the strong La Nina and the effects of Category 5 Tropical Cyclone Yasi have had devastating effects on large areas of GBR seagrass (Devlin et al., 2012b; McKenzie et al. 2015).

Port dredging may have severe but more short-term effects on seagrass in the GBR (although the potential for spoil dumping to damage seagrass is a hotly debated topic), such as at Hay Point (York et al. 2015), where turbidity from dredging is believed to have prevented seasonal re-establishment of deep-water seagrass, although recovery occurred after the dredging ceased.

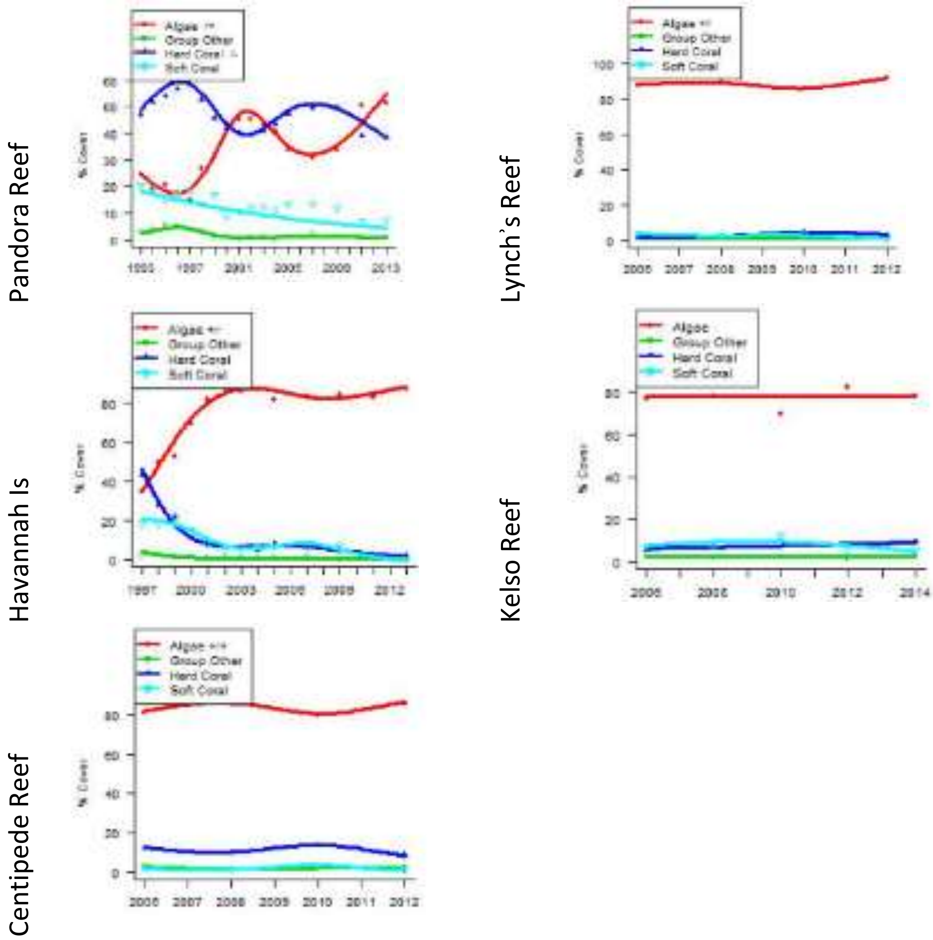

Figure 6. Coral cover, algal cover and other parameters for period 2005 – 2014 for LTMP reefs in the Burdekin marine region. Figure modified from figure and data in AIMS (2015).

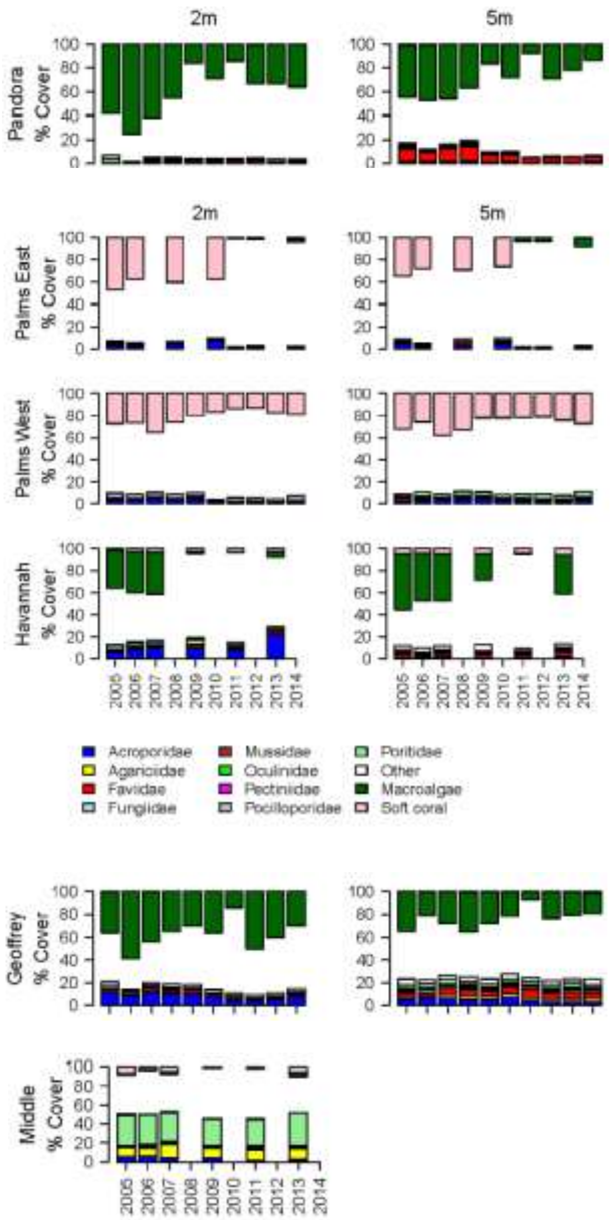

Figure 7. Coral cover by family and by depth over period 2005 – 2014 for MMP reefs in the Burdekin marine area. The colour pink shows soft coral, the white other cover, the dark green macroalgal cover while all other colours are for hard coral of different families. Cover estimates indicate regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Source: Modified from data and figure in Thompson et al. (2014).

6 Debate over prioritisation for management across the GBR among regions, catchments, land uses and industries, and within catchments

Assessments to prioritise catchments and land uses for management on the GBR catchment began formally with the prioritisation by Greiner et al. (2003, 2005), based on a multi-criteria analysis tool for assessing the relative impact of diffuse-source pollution from the river basins draining into the GBR lagoon. The assessment integrated biophysical and ecological data with socio-economic information pertaining to non-point-source pollution and (potential) pollutant impact. The analysis generated scores for each river basin against four criteria, thus profiling the basins and enabling prioritization of management alternatives between and within basins. The criteria were: (1) (Potential) ecological impact of diffuse-source pollution; (2) (Potential) social impact of pollution prevention; (3) (Potential) economic impact of pollution; (4) Development pressures.

The analysis was advanced in that it included offshore GBR asset values, and the capacity of catchment communities to manage natural resources, which were factors not included in subsequent assessments until the most recent Water Quality Improvement Plans (WQIPs) (e.g., Burnett Mary NRM 2015). No single ranking of basin priorities across the 35 GBR Basins was produced due to conceptual difficulties in combining scores from the four criteria. All subsequent prioritisation analyses followed similar methodology to Greiner et al. (2003, 2005).

In response to the needs of the Australian Government's Reef Rescue funding allocation in 2008, a new multiple-criteria prioritisation assessment was carried out (Cotsell et al. 2009), which was further developed at a workshop at which scientists supplied weights for each criterion to generate a multiple-criteria score for each NRM region. The outcomes of the workshop were presented at stakeholder forums to inform proposal development for Reef Rescue Water Quality Grants and Partnerships funds. This prioritisation process proved useful for the logical and transparent treatment of a wide range of data sets and facilitated the structured engagement of Reef Rescue implementers with Reef scientists and stakeholders. However, a major limitation of the decision support model was the lack of adequate data sets for solvability criteria.

In 2009 the Queensland Government introduced regulatory action to ensure management of the catchments delivering the greatest impact on the GBR. The Barrier Reef Protection Amendment Act (2009) attempted to ensure that farmers adopt management practices that reduce the levels of farm pesticides, fertiliser nutrients and sediment harming the GBR (see Brodie et al. 2012a). It was announced that the package of regulations aimed to reduce pesticide and fertiliser pollution of Reef waters by 50% by 2013 and sediment pollution by 20% by 2020. These "targets" were based on estimates of the quantity of land-based contaminants discharged from the region, the proximity of the catchment to vulnerable reef ecosystems, the existing condition of the reef ecosystems and the nature of the industries contributing contaminant loads. The targets derived from another prioritisation MCA based on a relative risk assessment, but restricted to pre-determined regions of the GBR – the Wet Tropics, the Burdekin and the Mackay Whitsunday regions (Brodie et al. 2009b; Brodie and Waterhouse 2009). In this analysis it was recognised that previous prioritisations of management response between different pollutants, different land uses/industries and different regions using MCA methods were valuable but had been carried out with limited input data (Cotsell et al., 2009; Greiner et al., 2005). More sophisticated analyses with better input data were needed to confidently prioritise between pollutants. The study of Waterhouse et al. (2012), which extended the analysis of Brodie and Waterhouse (2009) to include five regions of the GBR (Wet Tropics, Burdekin, Mackay Whitsunday, Fitzroy and Burnett-Mary but excluding Cape York), aimed to address previous assessment deficiencies by using improved input data and assessing the reliability of the data.

The assessment showed that the Wet Tropics and Mackay Whitsunday regions rank the highest priority (ranked high), with Burdekin and Fitzroy catchments relatively high priority (medium– high) and the Burnett Mary catchments of moderate priority in terms of the contribution and influence of land-based pollutants (Waterhouse et al. 2012). This assessment concurs with several principles of the current understanding of priority contaminants and land uses in the GBR:

1. Sugar cane and horticultural land uses that generate substantial runoff of DIN and PSII herbicides are dominant in the coastal areas of the Wet Tropics, Burdekin, Mackay-Whitsunday and Burnett-Mary catchments.
2. The predominantly coastal location of intensive agriculture in the GBR catchment results in efficient delivery of contaminants to the GBR.
3. Many reefs are located close to the coast in the northern parts of the GBR, particularly in the Wet Tropics, while most southern reefs are located further offshore.
4. The assessment reflects the importance of dry tropics grazing activities and the contribution of sediment by erosion to receiving waters. A large proportion of the reefs in the dominant grazing areas of the Fitzroy and Burdekin catchments are located further offshore and thus may present a lower risk. However, suspended-sediment risk to other important GBR ecosystems such as seagrass beds has not been included in this assessment; if this was done, the importance of the Burdekin and Fitzroy regions might be enhanced.

To assist in the development of Reef Plan 2013 and the SCS of 2013, a risk-assessment method was developed and applied (Brodie et al. 2013a, 2013b; Waterhouse et al. 2013). It aimed to inform policy makers and catchment managers on the land-based pollutants of greatest risk to the health of coral reefs and seagrass meadows. The approach used a combination of qualitative and semi-quantitative information about the influence on these ecosystems of individual catchments, in the 6 relevant natural resource management (NRM) regions. The method used a multiple-criteria approach with the application of a spatial tool, Multi-Criteria Analysis Shell for Spatial Decision Support (MCAS-S). The combined assessment of water quality variables was used to identify the areas where coral reefs and seagrass are at highest relative risk of impact of degraded water quality in the GBR. The relative risk was estimated from the areas of coral reefs and seagrass meadows exposed to a combination of defined pollutant thresholds (observed or modelled). The results indicated that the risk was greatest for coral reefs in the Fitzroy and Mackay Whitsunday regions, and for seagrass in the Burdekin and Fitzroy regions. The combined assessment of these results with inclusion of information on end-of-catchment pollutant loads allowed conclusions to be drawn about the overall risk of pollutants to the GBR (which differ from the assessment when loads are not considered). In summary, from this overall assessment, the greatest risk to coral reefs and seagrass meadows is in the Wet Tropics region, followed by the Fitzroy and Burdekin regions.

A similar analysis with additional data layers was carried out during 2013-14 to support and inform discussion and decisions on funding priorities for investment, particularly through Reef Water Quality Grants (part of the Australian Government Reef Programme) (Barson et al. 2014). The results showed that investments in improving practices in the sugar cane and grazing industries could be expected to give the biggest water quality improvements. In the Wet Tropics NRM region the priority is to improve the management of nutrients (fertiliser use) in the Johnstone, Russell-Mulgrave, Tully, Herbert and Daintree Basins, and in improving herbicide management in the Herbert and Johnstone catchments. In the Burdekin NRM region, investments in improved cane nutrient and herbicide management are expected to give the biggest returns in the Haughton catchment. In the Mackay Whitsunday NRM region, investment in improving cane herbicide management practices is likely to deliver the biggest water quality improvements. For the grazing industry, the biggest returns on investment in reducing sediment loss will come from the Burdekin and Fitzroy catchments.

The prioritisations from Greiner et al. (2005) through to Barson et al. (2014) have given generally consistent results with priorities given to fertiliser and pesticide management in the sugarcane cultivation industry and erosion management in the beef grazing industry. However, confusion is still present in the government response to these priorities, through misinterpretation of the science and caveats in the results, such that the Queensland election commitment by the incoming Palaszczuk government (see Section 4.7.1.1 above), as an addition to the LTSP, stated:

> *"Reduce total suspended sediment run-off by up to 50% in key catchments such as the Wet Tropics and the Burdekin by 2025."*

Unfortunately, the Wet Tropics has never been seen as a priority region for sediment (erosion) management, although it may be for particulate nutrient management (which is also related to erosion rates). The catchments that are identified for sediment management in our latest analyses are actually the Mary, Fitzroy, Burdekin and Normanby, none of which are in the Wet Tropics (Brodie et al. 2013b).

One important issue is still not resolved in the current prioritisation assessments. The current prioritisation methodologies focus on identifying the most impacted marine environments and correlating them to investment in management actions within the most disturbed sections of the GBR catchment. As a result, the limited resources for management action change have been invested in Reef regions with the largest disturbance in an effort to arrest the decline in Reef water quality. Through this approach only the most degraded sections of the Reef (e.g., central GBR) receive large-scale funding while the areas of the Reef still in good condition (e.g., Cape York) receive little funding. As a result, perhaps, the current investment prioritisation method, coupled to the current level of on-ground investment and lack of commitment to meaningful Reef regulation, is doing little to arrest the decline in GBR water quality and ecosystem health (Brodie and Waterhouse 2012; GBRMPA 2014; Hughes et al. 2015).

Given this issue of too little funding ploughed into areas of high catchment degradation with little hope of major success in reducing pollutant loads, a triage approach through a conservation biology framework is now being suggested as an important addition to current management thinking (Will Higham pers. com.). Through this approach, regions such as Cape York, where we know reefs, seagrass and dugong populations are in good condition (Coppo et al. 2016), would be prioritised for protective management – for example, agricultural development to only be allowed with immediate adoption of A class practices. Similarly, given the good status of seagrass and dugongs in Hervey Bay (Coppo et al. 2014), the river most influencing Hervey Bay, the Mary (Burnett Mary NRM Group 2015; Brodie and Lewis 2014; Waterhouse et al. 2014), would be prioritised for erosion management (as it already is following these principles in the Burnett-Mary WQIP – Burnett Mary NRM Group 2015) (Brodie and Pearson 2016).

7 Consensus

The above issues have never been fully resolved, as might be expected for such complex systems and processes, and continue to be debated vigorously. However, various syntheses, scientific consensus statements, review papers, books and technical reviews have attempted to resolve the issues over the last 25 years such that management could be prioritised on the basis of the best understanding of the issue at the time (e.g., Williams 2002; Furnas 2003; Brodie et al. 2001b; Productivity Commission 2003; Haynes et al. 2001). The progress towards a consensus view by 2003 allowed development of the Reef Water Quality Protection Plan ("Reef Plan”), described below.

The first scientific consensus statement on the issue (Williams et al. 2001) concluded that:

- available evidence indicates that post-European land use has significantly increased runoff and sediment, associated nutrient and contaminant delivery to near-shore regions of the GBRHWA;
- runoff has had clear detrimental impacts on freshwater aquatic systems; and
- there is significant risk that this impact is currently or may in future damage areas of high exposure along the wet tropical and central Queensland coasts of the GBRWHA, and there is a continued urgency to work towards a reduction in the runoff of sediments, nutrients, herbicides and other pollutants into the Great Barrier Reef World Heritage Area.

Similarly, the Productivity Commission report (Productivity Commission 2003) concluded that:

- Water quality in rivers entering the Great Barrier Reef (GBR) lagoon has declined because of diffuse pollutants, especially sediments, nutrients and chemicals from cropping and grazing lands in relatively small areas of the adjacent catchments. This diffuse pollution threatens inshore reefs and associated ecosystems.
- Because of the World Heritage values at risk, a strategy to identify, prioritise and manage risks is warranted, notwithstanding remaining scientific uncertainty about the condition of reefs and the effectiveness of remedial actions.
- Existing water quality policies largely ignore diffuse pollution and involve prescriptive end-of-pipe controls. Prescription is not the answer. Because of the complexity, heterogeneity and dispersion of the diffuse sources, and the inability to monitor them, governments cannot prescribe land management practices that are both viable and cost-effective.
- Solutions will have to be built up from local knowledge and insights, within a general framework set by the Commonwealth and Queensland Governments.
- Some primary producers (from each industry) have already demonstrated that it is possible and viable to reduce land and water degradation on their own lands. The challenge is for these practices to be more widely adopted or adapted.
- No single solution will control diffuse pollution entering the GBR lagoon. Various combinations of measures — tailored to particular land uses, locations, and pollutants — will be necessary, giving land users flexibility to choose abatement actions best suited to their property.
- Local groups have an important role in designing and delivering programs and monitoring outcomes, but serious questions remain about the structure, transparency and accountability of proposed regional groups.

- Regional groups should not create an additional layer of complexity but instead be part of a simplified approach that is integrated with the actions of other parties, notably the Commonwealth and Queensland Governments.
- Improving downstream water quality in rivers and estuaries flowing into the GBR lagoon will generate benefits apart from reducing the threat to the Reef. But zero discharge is unnecessary and, if possible at all, would be at prohibitive cost.

8 Science to management – Reef Plan 2003

By 2001, the large amount of research and monitoring carried out over the previous 20 year on the GBR, and its synthesis into a coherent body of knowledge through the processes described above, was sufficient to, in the appropriate political and governance environment (see below), begin formulating a plan to manage terrestrial pollutant runoff to the GBR.

However, it still required the intervention of Senator Robert Hill, Minister for the Environment (and Heritage) 1996 – 2001 in the Australian Government and a powerful political figure within the Liberal Party, to initiate action. His decision, at a GBR Ministerial Council meeting in Cairns in early 2001, to ask for a report setting out calculated targets for pollutant reduction for the rivers of the GBR Catchment led to a group, which I led, being tasked with the target-setting process at the meeting. The group proceeded over the next few months to prepare the target report (Great Barrier Reef Catchment Water Quality Action Plan) (Brodie et al. 2001a) with the best available knowledge at that time. Rod Welford was the Minister for Environment & Heritage and Minister for Natural Resources in the Queensland Government from 1998 to 2001 and also contributed greatly to the subsequent agreement between the Australian and Queensland Governments over Reef Plan 2003.

This critical conjunction of the Federal Coalition Government and Queensland Labor Government in 2001 at the time that much of the science had come together (see above), and Minsters for the Environment in both governments convinced by the science and willing to act, was crucial in the formulation of the Reef Water Quality Protection Plan (Reef Plan 2003). Following this Ministerial Council meeting in 2001, through the efforts of many government scientists and environmental and natural resource managers, agricultural industry, university and research agency scientists, conservation organisations, and of particular mention, Sheriden Morris (Director, Water Quality and Coastal Development, GBRMPA) and Helen Ringrose (Queensland Government), the Reef Plan was formalised in 2003.

9 From Plan to action

While Reef Plan 2003 was more about a “Plan to develop a Plan” than an actual implementable Plan for on-ground actions, its symbolic importance was huge and led to substantial funding in 2008 to implement on-ground actions (Brodie et al. 2012a). From 2001 to the present, our knowledge of the water quality issues of the GBR and potential solutions have improved greatly through research, evaluation and monitoring. To facilitate use of this new knowledge, two further Scientific Consensus Statements were prepared in 2008 (Brodie et al. 2008a, b) and in 2013 (Brodie et al. 2013a). The statements were used to produce revised Reef Plans (Reef Plan 2009 and Reef Plan 2013). The improved knowledge brought together in 2008 can be summarised in the conclusions of the 2008 SCS as follows:

- Water discharged from rivers to the GBR continues to be of poor quality in many locations.
- Land derived contaminants, including suspended sediments, nutrients and pesticides are present in the GBR at concentrations likely to cause environmental harm.
- There is strengthened evidence of the causal relationship between water quality and coastal and marine ecosystem health.
- The health of freshwater ecosystems is impaired by agricultural land use, hydrological change, riparian degradation and weed infestation.
- Current management interventions are not effectively solving the problem.
- Climate change and major land use change will have confounding influences on GBR health.
- Effective science coordination to collate, synthesise and integrate disparate knowledge across disciplines is urgently needed.

Similarly, the 2013 SCS concluded:

- Overall the overarching consensus is that key Great Barrier Reef ecosystems are showing declining trends in condition due to continuing poor water quality, cumulative impacts of climate change and increasing intensity of extreme events (Brodie et al. 2013a).
- The decline of marine water quality associated with terrestrial runoff 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 (Schaffelke et al. 2013).
- The greatest water quality risks to the Great Barrier Reef are from nitrogen discharge, associated with crown-of-thorns starfish outbreaks and their destructive effects on coral reefs, and fine sediment discharge which reduces the light available to seagrass ecosystems and inshore coral reefs. Pesticides pose a risk to freshwater and some inshore and coastal habitats (Brodie et al. 2013b).
- Recent extreme weather—heavy rainfall, floods and tropical cyclones—have severely impacted marine water quality and Great Barrier Reef ecosystems. Climate change is predicted to increase the intensity of extreme weather events (Johnson et al. 2013).
- The main source of excess nutrients, fine sediments and pesticides from Great Barrier Reef catchments is diffuse source pollution from agriculture (Kroon et al. 2013).
- Improved land and agricultural management practices are proven to reduce the runoff of suspended sediment, nutrients and pesticides at the paddock scale (Thorburn et al. 2013b).

Following the allocation of large-scale funding from both the Australian and Queensland Governments in 2008 (Brodie et al. 2012a), on-ground management action improvements began in earnest. They included fencing in grazing lands to restrict cattle access to waterways and maintain pasture cover, and fertiliser management in sugarcane cultivation to match crop needs to fertiliser application rates to reduce fertiliser loss from the paddock to waterways. The success of these interventions and of Reef Plan in general have been addressed by Brodie and Pearson (2016).

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