

Shifting base-lines, declining coral cover, and the erosion of reef resilience: comment on Sweatman et al. (2011)

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Abstract Formal monitoring of the Great Barrier Reef was initiated in 1986 in response to the clear scientific evidence (and growing public concern) over the loss of corals caused by two protracted outbreaks of crown-of-thorns starfish, which began in 1962 and 1979. Using monitoring data from manta tows along and across the Great Barrier Reef, Sweatman et al. (Coral Reefs 30:521–531, 2011) show that coral cover after these outbreaks declined further from 28 to 22% between 1986 and 2004. Pointing to the current levels of protection of the Great Barrier Reef, they state that earlier estimates of losses of coral cover since the early 1960s have been exaggerated. However, the loss of close to one-quarter of the coral cover over the past two decades represents an

average loss of 0.34% cover per year across the whole GBR after 1986, which is very similar to previously reported rates of annual loss measured over a longer timeframe. The heaviest recent losses have occurred on inshore and mid-shelf reefs, which Sweatman et al. (Coral Reefs 30:521–531, 2011) attribute to a natural cycle of disturbance and recovery. But there has been very limited recovery. While coral cover has increased for short periods on some individual reefs, it has declined sharply on many more to produce the observed system-wide trend of declining cover. Close to 40% of coral cover on inner reefs has been lost since 1986. Of particular significance is the new evidence that coral cover has remained unchanged or declined further from a low 1986 baseline in 28 out of 29 sub-regions of the Great Barrier Reef, indicating a gradual erosion of resilience that is impeding the capacity of this huge reef system to return towards its earlier condition. This result, and other clear evidence of widespread incremental degradation from overfishing, pollution, and climate change, calls for action rather than complacency or denial.

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In recent years, a wealth of new evidence has documented the slow degradation of the Great Barrier Reef since the mid-1800s. Stocks of pearl shell, sea cucumbers, turtles, and dugongs were heavily depleted by the first wave of industrial-scale fisheries (e.g., Jackson et al. 2001; Pandolfi et al. 2003), while changes on land following the arrival of Europeans and their livestock, and from land clearing and the development of fertilized cropping (e.g., sugarcane), have all increased the inflow of sediment and nutrients onto

nearshore reefs (McCulloch et al. 2003; McKergow et al. 2005a, b). More recently, coral cover was reduced on many reefs by three protracted large-scale population outbreaks of the crown-of-thorns starfish, *Acanthaster planci*, which began in 1962, 1979, and 1993 (Fabricius et al. 2010). Coral bleaching from elevated temperatures occurred at unprecedented scales in 1998 and in 2002 (Berkelmans et al. 2004). In 2004, the GBR Marine Park was rezoned in a decisive response to these unfavorable trajectories, to rebuild depleted fish stocks, repair distorted foodwebs, and to build the resilience of the reef in anticipation of future bouts of bleaching from inevitable climate change (Hughes et al. 2007; Olsson et al. 2008; McCook et al. 2010).

Sweatman et al. (2011) argue that coral cover on the Great Barrier Reef (GBR) has not changed appreciably, remaining the same in 22 out of 29 sub-regions since 1986. They challenge historical evidence of longer-term degradation of coral reefs and other ecosystems on the GBR (e.g., Jackson et al. 2001; Pandolfi et al. 2003), ignore most of the recent literature on the impacts of coastal runoff and coral bleaching, and claim that earlier studies on the loss of coral cover on the GBR (Bellwood et al. 2004; Bruno and Selig 2007) have exaggerated the decline. Yet, their own results show that between one-fifth and one-quarter of the coral cover present on the Great Barrier Reef in 1986 has been lost, with close to a 40% loss from inner reefs in just 19 years. Despite these losses, in most sub-regions of the GBR, at least one reef sampled between 1986 and 2004 exhibited an increase in coral cover, for an average period of 5–6 years, before declining again (see Table 1 in Sweatman et al. 2011). However, these short-term recoveries are not representative of the general trend. Among the 29 sub-regions examined by Sweatman et al. (2011), only one showed a small net increase in coral cover after 19 years. Therefore, this lack of recovery in all but one sub-region and the continued decline in coral cover on the GBR as a whole indicates a gradual erosion of resilience that is system-wide in scale, despite the temporary increases in coral cover exhibited by a small number of individual reefs.

The sample size of Sweatman et al.'s (2011) monitoring program is impressive, ranging from 72–189 reefs each year, so there is no reason to doubt the accuracy of their results. It is the interpretation and management context of these important findings that we believe are flawed. Sweatman et al. (2011) conclude by stating that coral cover on the GBR could decline in the future, if multiple disturbances are interspersed by incomplete recoveries, and if coral growth and recruitment are reduced by further disturbances. However, our interpretation of the available evidence is that this scenario is already well underway due to human-enhanced drivers of change (especially sediment and nutrient runoff, *Acanthaster* outbreaks, and rapid

climate change). We maintain that declining coral cover is not just some future possibility—it has been a reality for decades.

In this commentary, we first ask why does coral cover decline and question the focus and accuracy of Sweatman et al.'s (2011) attribution of just three sources of acute mortality as the cause of long-term loss of coral cover. Secondly, we examine the utility of older records of coral cover for correcting shifting base-lines, and we refute Sweatman et al.'s (2011) claim that earlier studies reported losses of corals that were 3-times higher than their more recent results. Finally, we examine whether Sweatman et al.'s (2011) dismissal of runoff as a cause of ecosystem degradation is credible and conclude that it is not.

Why does coral cover decline?

Sweatman et al. (2011) explain the decline of coral cover on the GBR as solely due to three types of acute disturbances: outbreaks of crown-of-thorns starfish, cyclones, and coral bleaching. While these are undoubtedly all important, the methodology and mode of observation used to attribute the declines in coral cover on individual sub-sectors to particular disturbance events is very weak.

Many studies of reef degradation such as Sweatman et al. (2011) place an emphasis on mortality in an attempt to explain what causes a population to decline, but this is only half of the demographic equation. While the importance of recruitment for coral recovery is widely appreciated, the potential contributions of recruitment failure and/or reductions in colony reproduction and growth to the long-term decline of coral cover are often ignored (but see, e.g., Richmond 1997; Hughes and Connell 1999). For example, massive *Porites* colonies sampled from 69 reefs along the GBR show a 13% decline in linear growth rates, and a 14% drop in calcification since 1990, with no precedent for any similar decline for at least the past 400 years (De'ath et al. 2009). Similarly, a gradual impairment of reproduction and recruitment on stressed inshore reefs is likely to have contributed to the long-term depletion of coral populations, as well as impeding their capacity to recover in recent years (Done et al. 2010; Hughes et al. 2010). In the Caribbean, recruitment has declined dramatically on many reefs over the past 30 years, and even the longest-lived corals have slowly declined due to a lack of larval replenishment (e.g., Hughes and Tanner 2000). On the Great Barrier Reef, declines in the number of fecund adult colonies translate directly into fewer recruits (Hughes et al. 2002). Instead of focussing on what killed the corals, we need to ask how have changes in multiple demographic processes (recruitment, growth, acute and chronic mortality) contributed to reductions in coral cover, in order to

gain a better understanding of why many coral assemblages are declining and are no longer resilient to recurrent disturbances. Monitoring coral cover alone cannot answer these important questions, without a better focus on processes.

Corals die or lose tissue from many agents, including competition, predation, disease, and a variety of chronic and acute disturbances such as storms, temperature and salinity stresses, and coastal runoff. On the Great Barrier Reef and elsewhere, the relative importance of each of these varies hugely, both in space and in time. Of the three acute mortality agents they highlight, Sweatman et al.'s (2011) analysis of damage by crown-of-thorns starfish is arguably the most convincing, because the densities of *Acanthaster* on each reef were measured using the same manta tows. Nonetheless, counts of *Acanthaster* from manta tows in turbid inshore reefs are likely to be unreliable. Consequently, the finding by Sweatman et al. (2011) that 6 of the 8 inner-most sub-regions of the Great Barrier Reef were significantly affected by crown-of-thorns outbreaks is surprising, because these outbreaks are much more prevalent on mid-shelf reefs (e.g., Moran 1986).

Coral declines ascribed to cyclones by Sweatman et al. (2011) were inferred indirectly from weather data and tracks of hurricanes. This approach is questionable, because proximity to a cyclone does not translate necessarily to high rates of loss. A strong, category 5 cyclone can have destructive winds of 200 km per hour up to 200 km from its center, while a weak category 1 or 2 cyclone can pass over a reef yet cause little damage. For example, 17 cyclones passed within 200 km of Heron Island in 11 of the 30 years between 1962 and 1992, but depending on their strength, direction, and duration, only 5 of them resulted in elevated rates of coral mortality (Connell et al. 1997). Sweatman et al. (2011) do not explain how they differentiated weak versus strong cyclones, or what width of cyclone track they adopted to attribute loss of coral cover to storms.

Similarly, it is unclear how maps of widespread bleached reefs from aerial surveys along the Great Barrier Reef in 1998 and 2002 were used by Sweatman et al. (2011) to infer that bleaching had caused significant losses of corals at only two out of the 29 sub-regions they studied. For example, aerial surveys of the GBR documented widespread and severe bleaching on mid-shelf reefs near the coastal city of Cairns in early 1998 (Berkelmans and Oliver 1999), while in situ surveys on four adjoining reefs (Green, Michaelmas, Upolu and Moore) confirmed extensive bleaching mortality on the flat, crest, and upper slopes (T. P. Hughes, pers. obs.). Two of these reefs were surveyed by Sweatman et al. (2011), yet they attributed the decline of corals in this sub-region solely to *Acanthaster* and cyclones. We suggest that this disparity arises from the

limitations of manta tows, which cannot accurately distinguish mortality events caused by bleaching, disease, or predation by crown-of-thorns starfish, especially when observations are taken months afterwards. All three sources of mortality leave behind intact dead coral skeletons that slowly degrade and break up, obscuring the causes of mortality. If a cyclone occurs after bleaching, disease, or starfish predation, but prior to the next manta tow census, the presence of toppled skeletons and coral rubble does not necessarily signify that the corals were killed by physical disturbance, since many of them may have already been dead for months or years.

Acute mortality events, while important, are only part of the reason why corals die or coral cover declines. For example, only one-third of the mortality of corals at exposed sites on Heron Island over 30 years was incurred in years with cyclones. The rest died unspectacularly from background levels of predation, competition, and physiological stress (Connell et al. 1997; Hughes and Connell 1999). The trajectories of coral cover in many GBR sub-regions reported by Sweatman et al. (2011) also indicate that a more protracted loss of coral cover commonly occurs. For inner reefs, overgrowth by macro-algae, reduced capacity for growth due to increasingly turbid water, smothering by sediment and an increased incidence of coral diseases and bioerosion are important sources of chronic mortality and recruitment failure (e.g., Fabricius 2005; Birrell et al. 2008; Fabricius 2011) that cannot be quantified by manta tows.

Harnessing long-term data

Although they present no data from before 1986, Sweatman et al. (2011) argue that longer-term losses of corals and other environmental changes on the Great Barrier Reef have been exaggerated. The title of their paper "Assessing loss of coral cover on Australia's Great Barrier Reef over two decades, with implications for longer-term trends" suggests that if their results (from 1986 onwards) are different from longer-term analyses, then the historical records must be wrong. This is not a logical argument. In the Caribbean for example, relatively small losses of corals have occurred in recent decades, following a much greater decline in the late 1970s and through the 1980s (Gardener et al. 2003). Referring to the Great Barrier Reef as a paragon of resource management in both their Abstract and Introduction, Sweatman et al. (2011) infer that two studies reporting a long-term decline in coral cover on the GBR (Bellwood et al. 2004; Bruno and Selig 2007) must be incorrect. The reality, however, is that substantial environmental damage on the GBR occurred long before modern reef management was implemented (e.g., Jackson

et al. 2001; McCulloch et al. 2003; Pandolfi et al. 2003; Hughes et al. 2010), including ongoing losses of corals that began decades before formal monitoring was initiated in the mid-1980s (Bellwood et al. 2004; Bruno and Selig 2007; see also Fig. 1).

Furthermore, the success of management of the GBR Marine Park today is constrained by larger-scale drivers of change, including coastal development, pressure from international markets, and global climate change. Consequently, contemporary policies have so far demonstrably failed to redress coral bleaching or disease, to prevent outbreaks of *Acanthaster*, or, as yet, to significantly improve water quality. Although the Great Barrier Reef Marine Park was established in 1976, intensive management of the effects of fishing began much more recently with the Trawl Management Plan of 2000, followed by the GBR Zoning Plan of 2003, which was implemented in mid-2004 (Olsson et al. 2008; McCook et al. 2010). Concerted efforts at controlling terrestrial pollutant runoff began with the Reef Rescue Initiative of 2008 (Brodie et al. 2011). Meaningful management of climate change is beyond the capabilities of local authorities and has yet to begin in earnest at a national or global scale. It is therefore not surprising that coral cover on the Great Barrier Reef has declined in the face of many pressures, to similar values seen in coral reef regions like Indonesia and the Philippines (Bruno and Selig 2007).

Sweatman et al. (2011) dismiss historical information on corals from the Great Barrier Reef, stating incorrectly that “very few estimates of coral cover are available from anywhere in the GBR province prior to 1986”. On the contrary, the 1960s, 1970s, and early 1980s comprise a period of intensive study in Australia, including pioneering studies of coral population and community dynamics, phase-shifts, and large-scale zonation across the GBR (e.g., Connell 1973; Done 1982, 1992; Endean and Stablum 1973, 1975). A major focus was the impacts of disturbance from cyclones and outbreaks of crown-of-thorns starfish, and the capacity of reefs to recover afterwards (e.g., Pearson 1974, 1981). This period was a highly productive one for a large community of researchers, including Joe Connell, Terry Done, Bob Endean, Bob Pearson, Michel Pichon, Charlie Veron, Carden Wallace, and many others. While these older studies were of course never designed to sample the entire GBR, they nonetheless recorded coral cover from every year of this 25-year interval (Bellwood et al. 2004), and their spatial coverage was substantial. For example, three comprehensive studies by Pearson (1974), Endean and Stablum (1973), and Kenchington and Morton (1976) surveyed 64 individual reefs between 1970 and 1974. This body of work is readily accessible in mainstream journals, including *Science*, *Ecological Monographs*, and *Coral Reefs* (from 1982 onwards) and is still

heavily cited. Rather than denying its existence, it seems reasonable to interrogate this peer-reviewed literature, to assess the condition of the GBR described by experts for a quarter of a century before the more recent advent of formal monitoring programs. Consequently, Bellwood et al. (2004) and Bruno and Selig (2007) used these older studies to compile >100 records of coral cover from the GBR between 1962 and 1986. Bellwood et al. (2004) presented a scatter plot of all available records from 1963 to 2004, while Bruno and Selig (2007) undertook a more formal meta-analysis of changes in coral cover on the GBR and in 9 other Indo-Pacific subregions.

It is important to recognize that an unknown amount of variance in any such compilation or meta-analysis arises from differences in methodologies among different studies, and from a range of potential biases in the selection of sites (Hughes et al. 2010). This is an issue for modern data too. For example, coral cover in the manta tows reported by Sweatman et al. (2011) differs from the amounts recorded for the same period in repeatedly measured video transects in deeper water (6–9 m depths). Videos have now replaced manta tows as the primary methodology used in contemporary monitoring on the Great Barrier Reef (Osborne et al. 2011), whereas the older literature on corals is based on fixed or random line transects and photo-quadrats. In many respects, these are much more accurate than manta tows, which rely on categorical estimates made by eye by snorkelers towed behind a small boat. Sweatman et al. (2011) acknowledge the limitations of manta tows for counting corals and *Acanthaster* on inshore, turbid reefs, which accounts for their limited sampling on the most intensely used reefs on the inner GBR. Their analysis of trends in coral cover is further complicated by the sampling design of their manta tow program, which is neither random nor repeated, because individual reefs are included in the dataset in some years but missing from others. They also concede that manta tows cannot measure species composition or count the number of recruits, whereas many of the older published studies do provide this critical information (e.g., Connell 1973; Done 1985). Following individual colonies through time in repeatedly censused plots (e.g., Connell 1973) also provides much more reliable information on why they die. To summarize, we contend that older studies from the GBR and elsewhere have enormous value, and they should not be ignored or dismissed.

So, what does the older literature tell us about coral cover in the past, and how does it compare to the more recent information from the Great Barrier Reef? The inference from Sweatman et al. (2011) is that coral cover in 1986, averaging 28%, is close to its historical level due to the initiation of multi-zone management in the early 1980s, and that the extent of any decline beforehand has been exaggerated. This is a classic example of a shifting

baseline, where the recent and current condition of coral cover on the GBR is assumed to be the same as it was long before formal monitoring began. As noted by Sweatman et al. (2011), their monitoring program was established in 1986 in response to public and scientific concerns over sharp drops in coral cover caused by repeated outbreaks of crown-of-thorns starfish that occurred from the early 1960s onwards. Therefore, it is clear that the declines they record over the past two decades were preceded by earlier losses prior to 1986 (e.g., Endean and Stablum 1973, 1975; Pearson 1974, 1981; Done 1985). Global assessments and meta-analysis from elsewhere in the Indo-Pacific and the Caribbean also support the generalization that coral cover was substantially higher in most regions 20–50 years ago than it has been in the past two decades (e.g., Gardener et al. 2003; Wilkinson 2008; Bruno and Selig 2007).

A frequency distribution of coral cover (Fig. 1) before 1980 shows a wide range of coral abundances on the Great Barrier Reef, with a similar number of reefs exhibiting

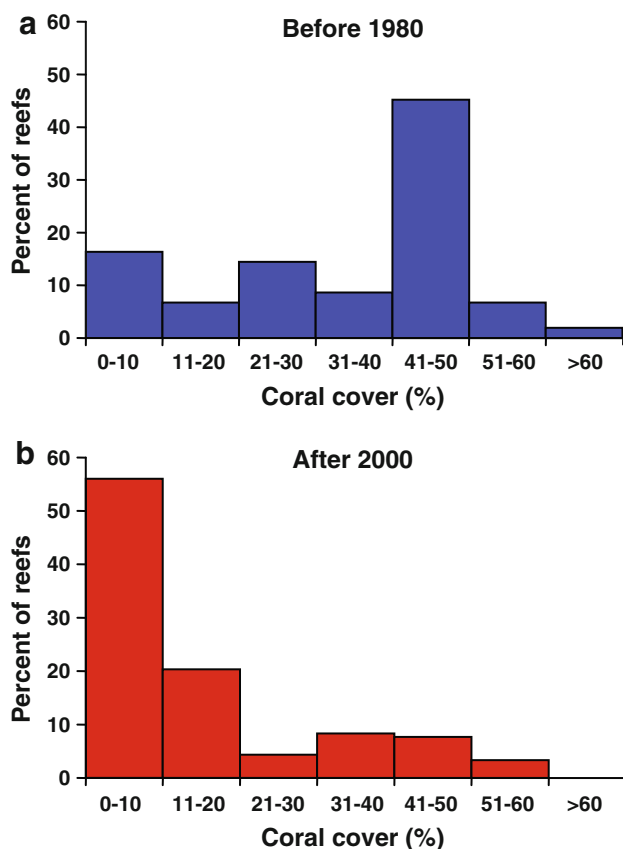


Fig. 1 Frequency distributions of coral cover recorded on the Great Barrier in (a) 1963–1980 and (b) 2000–2004. Based on original studies by Connell (1973), Connell et al. (1997), Done (1982, 1985), Endean and Stablum (1973), Fisk and Done (1985), Kenchington and Morton (1976), Pearson (1974), Sweatman et al. (2001, 2003), Veron (1978), Veron and Hudson (1978), Walsh et al. (1971). Re-drawn from Fig. 1 in Bellwood et al. (2004)

<10% and >50% average cover. Typically, these ends of the spectrum represent individual reefs before and after a cyclone or an outbreak of crown-of-thorns starfish. This broad range of coral covers recorded before 1980 does not support the notion that these earlier studies focussed only on a small subset of reefs that had unusually high cover, even by the standards of that time. The modal amount of corals recorded during this period was 41–50% cover. In contrast, since 2000, more than half of the reefs monitored on the Great Barrier Reef have <10% coral cover, and only a small number of reefs with cover >40% still persist (Fig. 1). A similar result for slightly different time periods (1960–1983 and 2000–2003) is presented by Bruno and Selig (2007). The most recent population outbreaks of *Acanthaster* affected 32% of the reefs monitored on the Great Barrier Reef between 1985 and 1997 (Lourey et al. 2000). Coral cover afterwards was 9% on outbreak reefs, compared to 28% on reefs that were unaffected by the most recent epidemics. This bimodal pattern is similar to the one revealed by analysis of the older records in Fig. 1, but with a significant decrease in the upper amounts. Based on a predator–prey model, Fabricius et al. (2010) estimate that coral cover today is 30–40% of its potential value (i.e., reduced by between three-fifths and two-thirds), a result that is entirely consistent with historical records (Bellwood et al. 2004; Bruno and Selig 2007; Fig. 1).

Sweatman et al.'s (2011) statement that their estimate of coral decline is 3-times lower than earlier reports is incorrect. The drop in cover they measured for the whole Great Barrier Reef was from 28.1% in 1986 to 21.7% in 2004, which represents an annual loss of 0.34% per year. Sweatman et al. (2011) do not report this observed rate, instead they emphasise a lower rate of loss based on a linear statistical model. The decline in coral cover reported earlier by Bellwood et al. (2004) was from an average of 38% coral cover in 1963 to 21% in 2000. Sweatman et al. (2011) exaggerate this longer-term decline by rounding the initial figure up to 40%, and the final one down to 20%. Using the correct figures, this represents an annual loss of 0.46%, which falls well within the upper 95% confidence limits of Sweatman et al.'s (2011) estimate for recent losses of cover on inner reefs (0.64% per annum), and just outside the upper confidence limit (0.45%) for midshelf reefs. There is certainly nothing like a 3-fold difference. Indeed, if Sweatman et al.'s (2011) statement was true, then a 1.02% loss of coral cover per year across the GBR since 1963 (i.e., 0.34% times 3) would have resulted in a catastrophic decline from 38% to zero (rather than to 21%) by 2000. Given that the threefold disparity claimed by Sweatman et al. (2011) does not exist, it follows that the explanation they give for how it arises is unnecessary and spurious. Rather, we maintain that all of these studies (Bellwood et al. 2004; Bruno and Selig 2007; Sweatman

et al. 2011, see also Fig. 1) reinforce the conclusion that corals on the inner- and mid-shelf GBR have been declining since at least the 1960s, by an annual rate of approximately 0.3–0.4% per annum, representing a loss of roughly half the coral cover over the past 50 years.

Water quality and reef decline

Sweatman et al. (2011) claim that exposure to runoff “affects only 5% of the reef area of the GBR”, and that “sedimentation and turbidity on inshore reefs may not have increased substantially since European settlement”. Both of these statements are wrong.

Of the circa 2800 individual reefs comprising the Great Barrier Reef, today an estimated 23% lie above thresholds in water clarity (<10 m Secchi disk depth) and chlorophyll concentration ($>0.45 \mu\text{g l}^{-1}$) that promote high levels of macroalgae and suppress species richness of hard and soft corals (DeVantier et al. 2006; De’ath and Fabricius 2010). Has that percentage increased in recent times? Skeletal band records from *Porites* since 1750 show a 5- to 10-fold increase in the delivery of sediments after 1870, when human settlement and land-use practices rapidly changed (McCulloch et al. 2003; Lewis et al. 2007). Recent water quality data from regional catchments, combined with information on land-use change, provide a second strand of evidence that riverine loads of sediment, nutrients, and pesticides are far higher now than in pre-European times. Sediment loads have increased by 5- to 10-fold, while total phosphorus and total nitrogen are 2–10 and 2–5 times higher (McKergow et al. 2005a, b; Brodie et al. 2008). During floods, satellite imagery shows that river plumes and associated phytoplankton blooms can cross the GBR lagoon into the Coral Sea, transporting dissolved matter to inner, middle and even outer reefs (Devlin and Brodie 2005; Devlin and Schaffelke 2009; Brodie et al. 2010, 2011). For the first time in the history of the GBR, biologically active concentrations of agricultural pesticides and herbicides have been measured during the wet season when rivers are running strongly, up to 60 km offshore (Brodie et al. 2008; Lewis et al. 2009).

The first Secchi disk records from the GBR were collected by the Royal Society expedition to the Low Isles, near Cairns, during the dry season in 1928, when average water clarity was recorded as 11 m. The contemporary average for the same months at the same location is only 5 m (Wolanski and Spagnol 2000). The intertidal coral assemblages recorded on the Lowe Isles in great detail in 1928–1929 have virtually disappeared (Bell and Elmetri 1995), consistent with De’ath and Fabricius’s (2010) threshold prediction. Similarly, many coastal and nearshore fringing reefs, recorded in landscape photographs from the

late nineteenth century onwards, show thriving *Acropora*-dominated assemblages that are now covered in mud (Wachenfeld 1997; Hughes et al. 2010). The link between nutrient runoff and outbreaks of crown-of-thorns starfish, a long-standing hypothesis (Brodie et al. 2005), has recently been strengthened by studies that combine historical records of floods that create nutrient pulses, experiments on feeding and development of starfish larvae, and population models that investigate the link between population explosions, larval development and food availability (Fabricius et al. 2010). Added nutrients also increase the productivity of macro-algae, consistent with the steep gradient in macroalgal abundance across the Great Barrier Reef and with the increased prevalence of phase-shifts from corals to fleshy seaweeds (e.g., Fabricius et al. 2005, Done et al. 2007; Wismer et al. 2009; De’ath and Fabricius 2010; Cheal et al. 2010). Coral diseases have recently been linked to water quality decline on the GBR (Haapkyla et al. 2011), as reported earlier from other coral reef systems (Bruno et al. 2003). Bioerosion of corals also increases as water quality declines (Le Grand and Fabricius 2011).

As a final comment, the study by Sweatman et al. (2011) highlights the potential for improving the design and focus of contemporary monitoring programs on coral reefs around the world. The decline in coral cover documented in their analysis begs the questions: why did it occur and what demographic processes are involved, how has the species composition changed, and what can we do to fix it? Unfortunately, the conventional focus on total coral cover and acute mortality events can only provide speculative responses, at best, to these critical questions. In a rapidly changing world, we recommend that monitoring programs innovate to measure processes such as recruitment and connectivity, survivorship and growth of corals, predation, herbivory and disease, and adapt their design to test hypotheses that can experimentally trial and improve management options (Hughes et al. 2007, 2010). Monitoring could become a much more powerful tool if it tested management options, using before and after comparisons with appropriate controls, rather than simply documenting the seemingly inexorable decline of coral cover.

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