

Effects of coral bleaching on coral habitats and associated fishes

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ABSTRACT

Coral reef ecosystems are particularly vulnerable to global climate change, owing to extreme environmental sensitivities and consequent bleaching of reef-building scleractinian corals. Severe coral bleaching often kills scleractinian coral, leading to longer-term declines in habitat and topographic complexity. The loss of corals, and corresponding changes in habitat quality, also has a significant effect on coral reef fishes. Coral loss has the greatest and most immediate effect on fishes that depend on live corals for food or shelter. Many coral-reef fishes that do not depend directly on live coral are nonetheless dependent on the topographic complexity provided by healthy coral growth. Sustained and ongoing climate change thus poses a significant threat to coral reef ecosystems and urgent action is required to minimize future effects of climate change and maximize resilience of coral reef ecosystems.

Key words: Climate change, coral reefs, disturbance, resilience

Introduction

Coral reefs are ecologically, economically and socially important ecosystems, renowned for their high biodiversity and productivity (e.g., Connell 1978; Hoegh-Guldberg 1999). However, coral reefs are being rapidly degraded throughout the world, contributing greatly to declines in biodiversity and productivity among coastal ecosystems (Worm *et al.* 2006). In the latest review on the status of the world's coral reefs, Wilkinson (2008) estimated that 19% of coral reefs have been essentially lost (whereby coral cover has declined by >90% and there is limited prospect of recovery), and a further 35% of reefs face a similar fate by 2050. The cause(s) of coral reef degradation vary greatly among geographic locations (e.g., Pandolfi *et al.* 2003). However, it is the areas closest to urban centres and large human populations that tend to exhibit the most pronounced degradation of coastal environments (Jackson *et al.* 2001; Pandolfi *et al.* 2003; Wilkinson 2008). A disproportionate number of coral reefs have been lost in east Africa, south-east Asia, and the central and southern Caribbean (Wilkinson *et al.* 2004), caused by chronic pollution, eutrophication, sedimentation, overfishing and/or destructive fishing practices. The long-term effects of anthropogenic disturbances are also being compounded by the increasingly large-scale impacts of global climate change (Hughes *et al.* 2003; Hoegh-Guldberg *et al.* 2007).

Climate change is widely regarded as the single greatest threat to the future of coral reef ecosystems, although the effects of climate change compound upon chronic long-term degradation of coral reef ecosystems (Hughes *et al.* 2003; West and Salm 2003; Hoegh-Guldberg *et al.* 2007; Wachenfeld *et al.* 2007). The long-term and cumulative effects of anthropogenic disturbances and increased fragmentation of coral reef habitats have greatly eroded reef resilience, making coral reef habitats much more susceptible to climate change (Hughes *et al.* 2003). Coral reefs that are isolated from direct anthropogenic disturbances and

coastal processes are not immune from the effects of global climate change (Graham *et al.* 2010), but it is likely that they will recover more rapidly in the aftermath of climatic disturbances (e.g., Sheppard *et al.* 2002).

Changes in atmospheric conditions due to anthropogenic climate change have direct effects on marine ecosystems, resulting in i) significant warming of shallow and surface waters (e.g., Gille 2002), and ii) increasing concentrations of CO₂ in ocean waters (Roessig *et al.* 2004). Water has a higher heat capacity than air, which means that increases in water temperatures lag behind that of atmospheric temperatures (Lough 2007). Even so, the average temperature at the surface of the ocean has increased 0.6°C over the last century, compared to 0.8°C for land-surface air temperature (Folland *et al.* 2002). Increasing temperatures are critically important because they bring baseline ocean temperatures much closer to the maximum thermal tolerances for reef organisms. Scleractinian corals especially, function very close to their upper thermal limit, such that bleaching may occur when sea temperatures exceed normal local limits by as little as 1.0°C (Jokiel and Coles 1990). A significant component of anthropogenically produced CO₂ (approximately one third, thus far) is also dissolved in the world's oceans (Roessig *et al.* 2004). Additional CO₂ dissolved in the ocean reacts with seawater to form weak carbonic acid, causing pH to decline and reducing the availability of dissolved carbonate ions required by many marine calcifying organisms (e.g. corals, other invertebrates, and coralline algae) to build their shells or skeletons (Orr *et al.* 2005).

Other climate related changes that will affect coral reef ecosystems include increasing severity of tropical cyclones (Madin and Connolly 2006), sea level rise (Short and Neckles 1999), and changes to ocean circulation and current patterns (Munday *et al.* 2008, 2009). Tropical cyclones are expected to become more intense in a warmer world (Webster *et al.* 2005), causing greater

damage to key habitat-forming species, such as corals and seagrasses (Madin and Connolly 2006). Sea level rise will lead to a redistribution of intertidal and shallow coastal habitats (Short and Neckles 1999). Changes to ocean circulation and current patterns will alter dispersal of marine larvae (Munday *et al.* 2008, 2009), and combined with reductions in vertical water mixing, will change the supply of nutrients and subsequent productivity of aquatic ecosystems. Climate related changes in environmental and habitat conditions will directly affect many coral reef organisms, resulting in changes in population dynamics, distribution and abundance (Munday *et al.* 2008), or for those species that are unable to adapt to changing conditions, climate change may lead to local or global extinction (e.g., Munday 2004). The most critical effects however (especially, in the shorter term), will result from changes in the biological and physical structure of coral reef habitats, formed by scleractinian (“hard”) corals (Pratchett *et al.* 2008).

Effects of climate change on corals

Coral reefs are considered to be among the most vulnerable ecosystems to global climate change (Walther *et al.* 2002), owing to sensitivities of habitat-forming corals to both increasing temperatures and ocean acidification (Hoegh-Guldberg *et al.* 2007). Ocean warming has already caused large-scale and severe episodes of coral bleaching throughout the world. In 1998, coral bleaching occurred in >50 countries, killing up to 90% of coral colonies. Australia was largely spared during the 1998 global mass bleaching; bleaching was conspicuous and widespread, but bleached corals mostly recovered (Wilkinson 2004) and overall mortality rates were generally low (Maynard *et al.* 2008; Anthony and Marshall 2009). However, large-scale bleaching has been observed in Australia and appears to be increasing in frequency and severity (Figure 1). On the Great Barrier Reef (GBR) major instances of coral bleaching have been recorded at fairly regular intervals

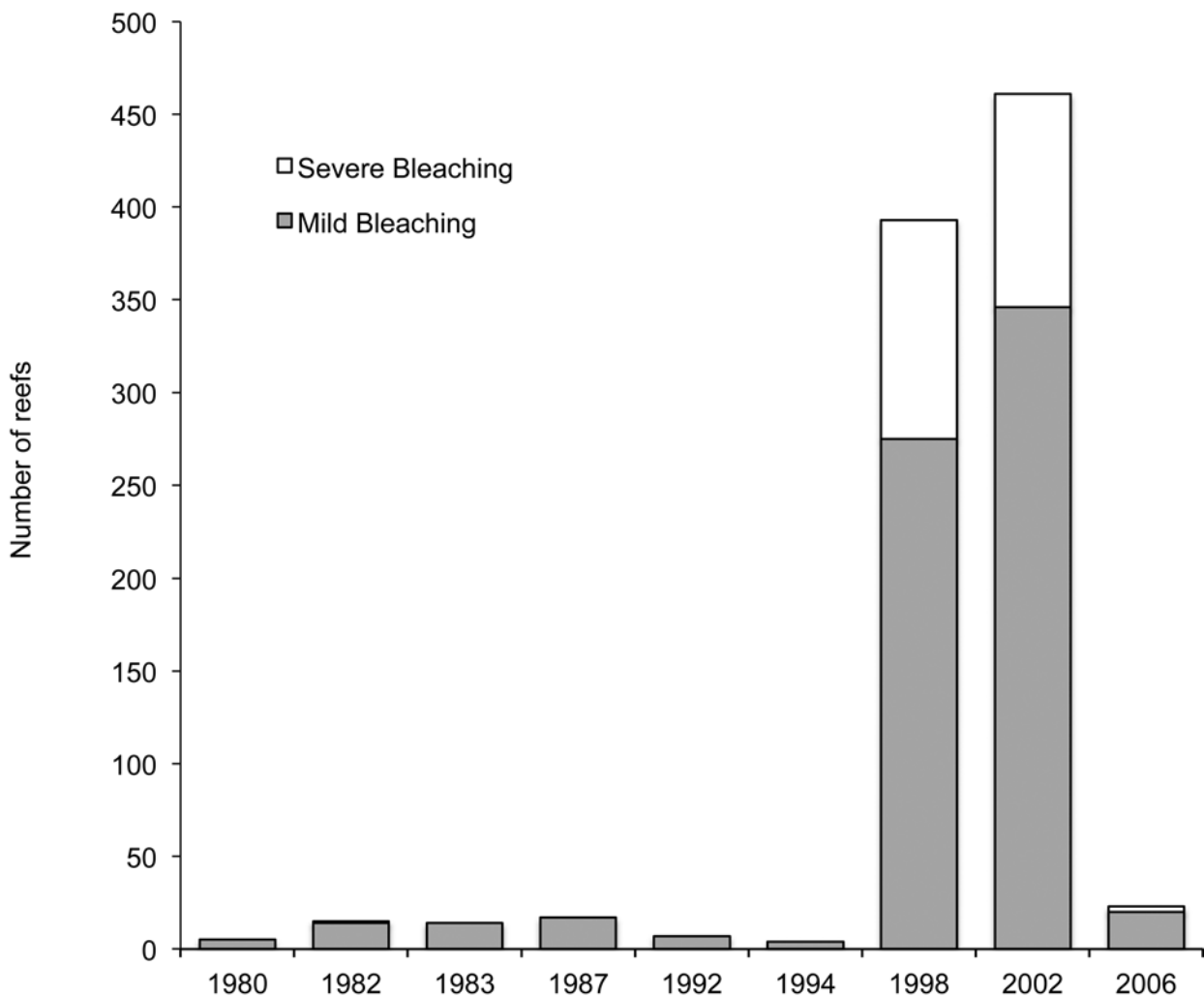


Figure 1. Recorded instances of mass coral bleaching on the Great Barrier Reef, since 1979. Seasonal “paling” of some coral colonies occurs in nearly all years, but specific instances of mass-bleaching, whereby multiple colonies and species are simultaneously affected has occurred 9 times (at intervals of 1-6 years). Data presented shows the number of reefs reported to have conspicuous evidence of bleaching, and the number of reefs with severe (>60%) bleaching. Variation in the number of reefs is partly due to limited spatial scale of surveys conducted in early years, but still there were a much higher proportion of severely bleached reefs in 1998 and 2002. For a complete account of data sources see Pratchett *et al.* (2011).

extending back to 1980, when conspicuous bleaching of common corals (mostly, *Acropora* and *Montipora*) was first noted at several isolated reefs between Townsville and Cairns (Oliver 1985). The most extensive and most severe bleaching episode to affect the GBR occurred in 2002 (Berkelmans *et al.* 2004; Maynard *et al.* 2008), corresponding with the highest sea surface temperatures (often >33°C) recorded on the GBR. During this event, bleaching was recorded at 54% of reefs surveyed across the length and breadth of the GBR (Berkelmans *et al.* 2004).

Given projected increases in sea surface temperatures, if corals are unable to acclimatize or adapt, coral bleaching events will become more frequent and more severe with time (Hoegh-Guldberg 1999; Donner *et al.* 2005). By 2050, most coral reefs are expected to be subject to annual thermal anomalies equivalent to those experienced in 1998 (Hoegh-Guldberg 1999), suggesting that mass bleaching will occur at intervals much less than the time required for corals (populations and communities) to recover from successive major bleaching events (Donner *et al.* 2005). Given strong taxonomic differences in susceptibility to bleaching (e.g., Baird and Marshall 2002), recurrent mass-bleaching is likely to alter the relative abundance of different coral species and growth forms (e.g., Riegl and Purkis 2009). However, the future state of coral communities will depend not only upon the differential bleaching susceptibilities of coral taxa (e.g., Baird and Marshall 2002), but also upon their capacity for recovery between successive bleaching events (Hughes *et al.* 2003). While several studies have documented increasing dominance of massive corals in coral communities subject to recurrent bleaching (Riegl and Purkis 2009), there also instances where highly susceptible corals (e.g., *Acropora*) have become even more dominant after bleaching events, due to their rapid recovery (e.g., Pratchett *et al.* 2009b). Ultimately, changes in community composition are likely to depend on the frequency, severity and spatial extent of future bleaching events. Effects of increasing temperature will also be further exacerbated by ocean acidification, which may reduce coral growth and population resilience (Hoegh-Guldberg *et al.* 2007).

The pH of oceans has declined by 0.1 pH unit since 1800, associated with increasing concentrations of H⁺ and HCO₃²⁻ (bicarbonate) ions and corresponding declines carbonate ion (CO₃⁻) concentrations. Declines of carbonate ions decrease the saturation state of calcium carbonate, particularly evident in more soluble calcite polymorphs such as aragonite and magnesium calcite (Kleypas and Langdon 2006). Reef building corals and crustose coralline algae construct skeletons from aragonite and magnesium calcite respectively, and are therefore particularly sensitive to changes in saturation states of calcium carbonate. Coral calcification rates are positively related to aragonite saturation state (Gattuso *et al.* 1998) and field data show that the growth rates of massive corals have declined by 15 – 20% over the past two decades (D'earth *et al.* 2009; Cooper *et al.* 2008). These declining growth rates of massive corals have not yet been irrefutably linked with changes in ocean carbonate chemistry, but may be the result of changes in ocean chemistry, water quality and/ or temperature effects (e.g., Anthony *et al.* 2008).

Effects of coral loss on fishes

Coral loss and associated changes in biological and physical structure of coral reef habitats have an important influence on the abundance and diversity of coral reef fishes (Wilson *et al.* 2006; Pratchett *et al.* 2008). Declines in coral cover generally lead to declines in the abundance of reef fishes, especially among fishes that rely on live coral for food, shelter and/ or recruitment (Kokita and Nakazono 2001; Munday 2004; Pratchett *et al.* 2004). Extensive coral loss may also result in declines in habitat and topographical complexity (Sheppard *et al.* 2002; Graham *et al.* 2007), which are critical for sustaining high diversity of reef fishes and other reef-associated organisms (Wilson *et al.* 2006; Pratchett *et al.* 2009). In the aftermath of extensive coral bleaching, the skeletons of dead corals are highly susceptible to biological and physical erosion (Glynn 1997; Hutchings 2011). Over time, coral skeletons of erect branching corals (e.g., *Acropora* and *Pocillopora*) break down into coral rubble (Sheppard *et al.* 2002; Graham *et al.* 2006), whereas more robust skeletons of massive corals (e.g., *Porites*) may become dislodged or gradually eroded *in situ* (Sheppard *et al.* 2002), potentially causing major declines in topographic relief (Pari *et al.* 2002). When coral loss is combined with structural collapse of reef habitats, up to 65% of reef fishes may experience declines in abundance (e.g. Jones *et al.* 2004; Graham *et al.* 2007). For fishes that are directly reliant on corals, sustained and ongoing declines in live coral cover may ultimately lead to local or global extinction. This is particularly so for highly specialist fishes that rely on only a limited suite of coral species (Munday 2004).

Corals as food

Scleractinian corals are an important source of food for at least 133 species (and 11 different families) of coral reef fishes (Cole *et al.* 2008; Brooker *et al.* 2010; Berumen and Rotjan 2010). The majority (69 species) of coral-feeding fishes are butterflyfishes (family Chaetodontidae), and more than half (69 out of 125 species) of all butterflyfishes feed at least in part on scleractinian corals (Allen *et al.* 1998; Cole *et al.* 2008; Froese and Pauly 2010). Coral feeding is an unusual feeding habit for most families of reef fishes. Aside from butterflyfishes, <5% of species (and often much less) within each of the major families of coral reef fishes are known to feed on corals; only 10 species of wrasses (family Labridae), 8 species of damselfishes (family Pomacentridae) and 8 species of parrotfishes (family Scaridae) are reported to feed on live corals (Cole *et al.* 2008). On the GBR, there are 27 species of fishes known to feed on coral, nearly all (70%) of which are butterflyfishes (Figure 2).

Of those fishes that do feed on hard corals, only 31% (41 of 133 spp.) are considered to be obligate coral feeders, meaning that they feed almost entirely (>80%) on live corals and their abundance is strongly linked to local coral abundance (Cole *et al.* 2008; Pratchett *et al.* 2008). Most importantly, these fishes exhibit rapid and dramatic declines in abundance following extensive coral depletion, such as that caused by mass coral bleaching, outbreaks of coral-eating crown-of-thorns starfish (*Acanthaster planci*),

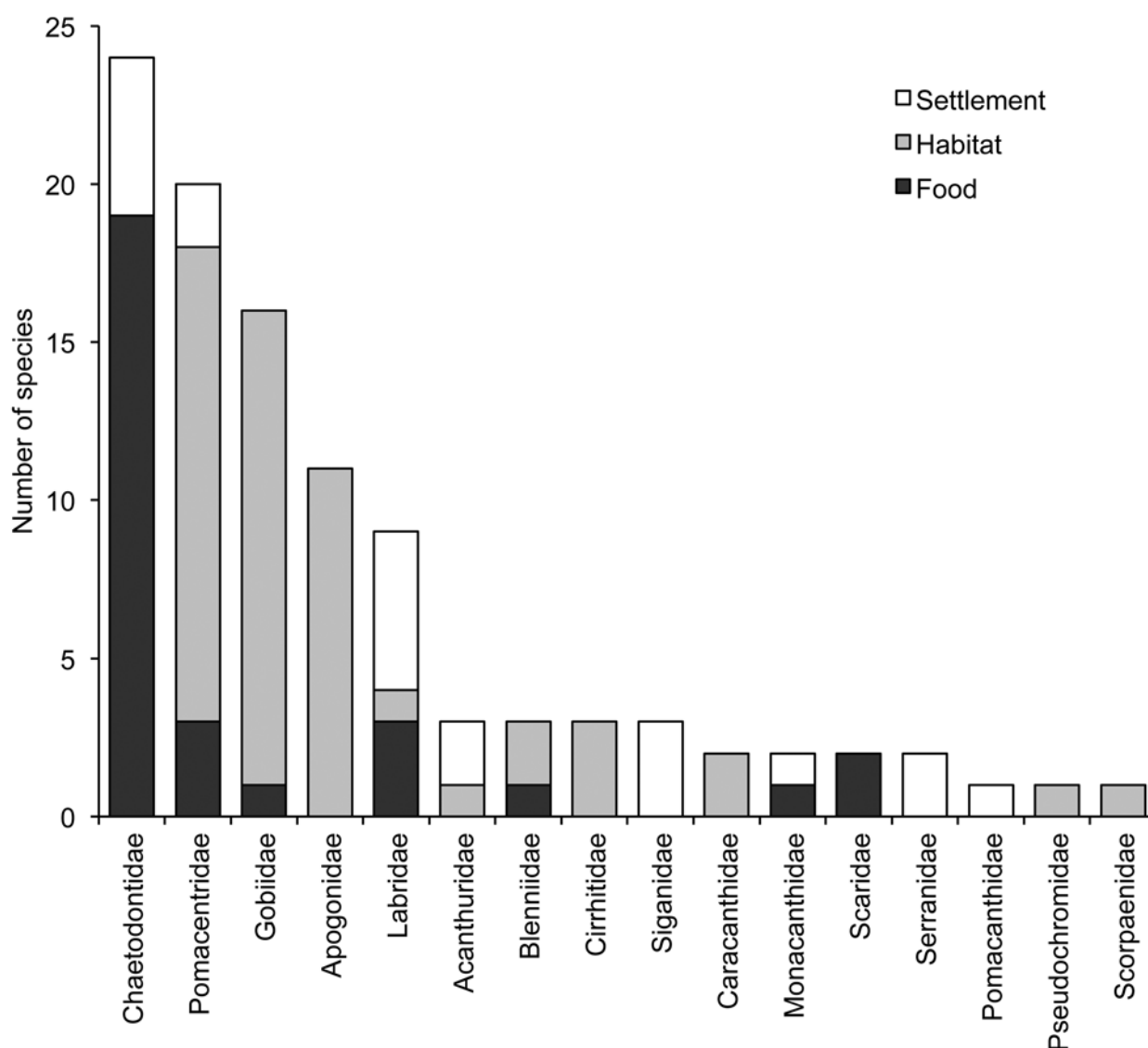


Figure 2. Coral-dependent fishes on the Great Barrier Reef, Australia; Number of species of fishes (by family) that rely on coral for either food or habitat.

or severe tropical storms (Kokito and Nakazono 2001; Sano 2004; Jones *et al.* 2004; Wilson *et al.* 2006; Pratchett *et al.* 2006, 2008). Moreover, obligate corallivores tend to have a highly specialized predator–prey relationship, and only consume a limited number of the available corals. This is considered evidence of the close co-evolution between the coral community and these fishes (Reese 1977), but also greatly increases their sensitivity to coral depletion.

Butterflyfishes exhibit particularly high levels of dietary specialization (Reese 1977; Hourigan *et al.* 1988; Tricas 1989; Cox 1994; Berumen *et al.* 2005; Pratchett 2005, 2007; Niedermüller *et al.* 2009). *Chaetodon trifascialis* is among the most specialized of all reef fishes and feeds almost exclusively on tabular *Acropora* throughout its geographical range (Reese 1981; Irons 1989; Alwany *et al.* 2003; Berumen and Pratchett 2006). Most coral-feeding fishes preferentially consume species from the genera *Acropora* and *Pocillopora* (Berumen *et al.* 2005; Pratchett 2005, 2007; Cole *et al.* 2008; Niedermüller *et al.* 2009), which are also those corals that are most vulnerable to bleaching (Marshall and Baird 2000). Accordingly, many

butterflyfishes (especially, *C. trifascialis*) have disappeared on reefs subject to severe coral bleaching. For less specialized coral feeders, which can switch feeding or generally feed on corals that are more resistant to disturbances, the effects of coral loss may be less severe or much more delayed (Pratchett *et al.* 2004). *Chaetodon lunulatus*, for example, consumes a much greater range of different corals (including massive *Porites*) compared to *C. trifascialis*. Following extensive coral bleaching in the central GBR in 2001–02, densities of *C. lunulatus* did not change for at least 2 years, but subsequently declined (Pratchett *et al.* 2006), possibly due to compromised fitness of individuals forced to feed on non-preferred corals.

Corals as shelter

Scleractinian corals are important contributors to both biological and physical habitat structure, such that any declines in the abundance or diversity of corals often have noticeable effects on local diversity and abundance of coral reef fishes (Jones *et al.* 200; Graham *et al.* 2006; Pratchett *et al.* 2008). Live coral habitat is important for many reef fishes

during their vulnerable juvenile stage (Dulyy *et al.* 2004; Feary *et al.* 2007a,b) as well as for many small-bodied adult reef fishes that live among the branches of live corals (Sale 1971; Munday and Jones 1998). On the Great Barrier Reef, 55 species of coral reef fishes live exclusively among the branches of live corals, mostly from the families Apogonidae, Cirrhitidae, Gobiidae, Pomacentridae and Scorpaenidae (Figure 2). Coral-dwelling fishes utilise complex branching corals with a strong preference to species from the family Pocilloporidae and the genus *Acropora* (Family Acroporidae). Accordingly, severe coral bleaching greatly reduces the availability of suitable coral hosts and leads to marked declines in abundance of coral-dwelling fishes (Jones *et al.* 2004; Munday *et al.* 2004a; Wilson *et al.* 2006).

Many coral-dwelling fishes vacate their coral hosts as soon as they become bleached, and few coral dwelling fishes will recruit to bleached coral hosts (e.g., Feary *et al.* 2007). This suggests that live corals are more important for these small coral-dwelling fishes than just the physical structure they provide (Booth and Beretta 2002). Most coral reef fishes are brightly coloured and stand out against pale and stark white bleached corals resulting in higher rates of predation as their visual camouflage is reduced to reef predators (Coker *et al.* 2009). While coral-dwelling fishes may sometimes persist on bleached corals, these fishes eventually move off in search of alternative healthy habitats if their host coral dies. Within one week of dying, algae and invertebrates (e.g., sponges, ascidians) will colonise the coral skeleton. At this stage, algae will start to take up the small gaps between the branches and reduce the available refuge spaces that fish can seek shelter within (Coker *et al.* 2009), such that fishes must move or face even higher risk of predation. The loss of live coral habitats means that healthy habitats will become a limited resource for these fishes and the migration of fishes to new habitats will create increased competition among coral-dwelling fishes. Fishes displaced by host coral mortality will have to join existing social groups of resident fishes on these colonies or, if unsuccessful, associate with less favourable habitats. Many coral-dwelling fishes have a tight social group and a strict size-based hierarchy (Forrester 1991; Wong *et al.* 2007), which will greatly limit opportunities to join established colonies on relatively unaffected coral hosts. Fishes may also have to travel considerable distances to find suitable habitats (especially, after extensive and widespread bleaching episodes), during which time they will be highly vulnerable to predation.

Many coral-reef fishes that do not feed on or live within live coral are nonetheless dependent on live coral, and may be negatively affected by significant coral loss (e.g., Jones *et al.* 2004). These include fishes that rely on corals to provide settlement cues and/or moderate key biological interactions, such as competition and predation. More research is still required to quantify the full range of fishes that rely on the biological or physical structure provided by corals, and might therefore be affected by comprehensive or severe coral loss. Thus far, effects of coral bleaching or coral loss on fishes of the GBR have been restricted to highly specialized coral-dependent species, including butterflyfishes, damselfishes and gobies (Munday *et al.* 1997; Booth and Beretta 2002; Pratchett *et al.* 2006). With

increased frequency or severity of bleaching episodes, it is likely that these fishes may become locally extinct (Munday 2004), but also a much greater range of different fishes are likely to be affected (e.g., Jones *et al.* 2004).

The future for coral reef structure and biodiversity

Resilience of corals

Sustained and ongoing increases in sea-surface temperatures, combined with declines in ocean pH, are expected to increasingly impact coral reefs in the future (Hoegh-Guldberg 1999; Hoegh-Guldberg *et al.* 2007). The future cover and species composition of scleractinian corals and the ecosystem they support, critically depend on not only their current resistance to environmental stress but also their ability to adjust to current rates of ocean warming and acidification in the future. Evidence for past adaptation in physiological and life history characteristics are apparent from correlations between bleaching thresholds and local environmental conditions of geographically isolated coral populations (Hughes *et al.* 2003; Barshis *et al.* 2010).

Coral communities can adapt to climate change through shifts in community composition, whereby coral assemblages become increasingly dominated by more tolerant species. (e.g., Hughes *et al.* 2003). Shifts in community species composition following bleaching have been widely documented (e.g., Loya *et al.* 2001). Recovery of reefs may occur through the regrowth of more resilient survivors (Loya *et al.* 2001) or through recruitment and recovery of fast growing, but often more sensitive species (Pratchett *et al.* 2008). Little is known about the effects of climate change on the re-colonisation potential of faster growing branching corals but coral bleaching can greatly affect growth rates of corals several years following bleaching event (e.g., Jones and Berkelmans 2010). Therefore, while climate change is expected to change the community composition of reef corals, it is not yet clear how impacts on growth and recruitment will affect the fitness of corals, ecological interactions within and among species, and hence their future species composition.

Natural selection through local adaptation is expected to increase the frequency of more tolerant individuals (genotypes) within populations. While increases in thermal tolerance of some coral populations have been observed following major bleaching events (e.g. Maynard *et al.* 2008), no study to date has directly linked differences in allelic frequencies to thermal tolerance among individual corals (Maynard *et al.* 2008). Bongaerts *et al.* (2010) found significant genetic structure of *Seriatopora hystrix* and their dinoflagellate symbionts within reefs but not among similar habitats on separate reefs. This result is consistent with ecological selection, a hypothesis that would be supported by physiological or molecular data to link coral-symbiont eco-types with fitness and stress tolerance. In the Caribbean, D'Croz and Mate (2004) found divergence in genetic structure and physiological tolerance between *P. damicornis* populations in cooler and warmer areas. Similarly, Edmunds (1994) found that rates of natural bleaching differed among

genotypes in *Montastrea annularis*. To maximise potential for adaptation (at community and population levels) it is important to maximise the abundance and diversity of corals, upon which selection can then operate. Herein, the resilience of reef corals to climate change will benefit greatly from effective local management, minimising anthropogenic disturbances that threaten corals (Hughes *et al.* 2003). Maintaining high gene flow or connectivity will also promote resilience and recovery from recurrent bleaching.

Resilience of fishes

Future increases in the frequency and/ or severity of coral bleaching events are inevitable, and as such, the persistence of coral-dependent fishes will depend (at least, in part) upon their own population and community resilience. A primary determinant of resilience will be the ecological versatility of coral-dependent fishes and their ability to use alternative resources if the abundance of preferred corals declines. While switching to alternative food sources or habitat types may enable some coral-dependent fishes to persist through periods of coral loss, increased use of non-preferred coral resources can have significant sub-lethal effects, such as declines in individual condition (Pratchett *et al.* 2006). Physiological condition is a major determinant of individual fitness in fishes and sub-lethal effects on body condition can have flow on effects to growth, reproductive success and survivorship (Jones and McCormick 2002; Munday *et al.* 2008). If use of non-preferred coral resources is continuous or prolonged, then sub-lethal effects may accumulate over time, gradually reducing survivorship and ultimately decreasing resilience.

Over longer time periods, the resilience of coral-dependent fishes will depend on population connectivity and especially, the proportion of new recruits originating from local or external sources. Populations that suffer severe declines in abundance will be slow to recover if recruitment is predominantly from local sources (i.e. high levels of self-recruitment). If however, significant levels of recruitment come from outside sources then recovery may occur via the long distance dispersal of recruits from unaffected populations (Hughes *et al.* 2005; Jones *et al.* 2009). Recent studies on population connectivity among coral reef fishes provide strong evidence for ecologically significant levels of self-recruitment (Swearer *et al.* 2002; Jones *et al.* 2009). These results suggest that for most fishes, declines in the local production of new larvae will greatly affect population viability. Climate change is expected to affect the connectivity of reef fish populations through changes in adult and larval biology and performance, and changes to larval supply and recruitment dynamics (reviewed by Munday *et al.* 2009). The net effect of these changes is likely to be an overall reduction in population connectivity (Munday *et al.* 2009), leading to increased reliance on local recruitment. In combination, these factors are likely to decrease the resilience of coral dependent fishes to the impacts of climate change.

As habitat perturbations become more frequent and more severe, it appears likely that highly specialised obligate coral feeders will be lost (Munday 2004). However, changes in the biological or physical structure of reef habitats may also benefit some fishes, such that there is no net decline in diversity or abundance of coral reef fishes (Bellwood *et al.* 2006; Berumen and Pratchett 2006). In general, fish communities in degraded post-bleaching habitats are characterised by dietary and habitat generalists (e.g., omnivores and detritivores), which replace coral-dependent specialists (Bellwood *et al.* 2006a; Graham *et al.* 2006). These post-bleaching fish assemblages may be fairly resilient to future disturbances but are nonetheless undesirable because the loss of entire functional groups (e.g., corallivores and herbivores) may have ramifications for recovery, productivity and ecosystem function (Bellwood *et al.* 2006a).

Conclusions

Global climate change is being caused by anthropogenic forcing of the climate system (Houghton *et al.* 2001), and not only are atmospheric concentrations of greenhouse gases rising, but the rate is accelerating (e.g. Canadell *et al.* 2007) due to increases both per capita and emissions and population growth (Houghton *et al.* 2001). As a consequence, even if climatic impacts are not yet apparent (or have had minor influence compared to other more direct anthropogenic disturbances) the effects of global climate change on ecosystems, communities and species will become increasingly important in the coming decades. Australia's Great Barrier Reef and other important reef ecosystems (Ningaloo and many offshore coral reef systems) have so far been spared from devastating effects of climate change that have already been witnessed on reefs in the Indian Ocean and Caribbean. However, sustained and ongoing climate change will cause increasing changes in abundance and community composition of corals and fishes on Australian coral reefs. It is important therefore, to act now to maximize resilience of reef organisms and ecosystems.

Urgent action is required to minimise global greenhouse gas emissions and thereby reduce longer-term climatic impacts on coral reef ecosystems. However, drastic reductions in emission, even if they are implemented immediately, will not guarantee the persistence of ecosystems, communities or species. Reductions in global greenhouse gas emissions will prevent extreme changes in environmental conditions and reduce rates of change to which species must adapt in order to survive. In the short-term, management must be focussed on minimising all other sources of anthropogenic interference coastal ecosystems that exacerbate vulnerability to climate change (Hughes *et al.* 2003).

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