Designer reefs and coral probiotics; great concepts but are they good practice?

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It has been estimated that over 500 million people as well as industries worth billions of dollars (including tourism and fisheries) depend on healthy reef ecosystems (Cesar 2000). As such, the continued decline in coral cover and shifts in community composition which are being observed on a global scale are an extremely worrying trend from economic, social and ecological perspectives. The overarching effects of anthropogenic stresses, including those related to climate change, are unarguably responsible for the recent unprecedented declines. However, understanding how these may interact with natural stresses in regard to their effect on corals is difficult, and in turn makes it difficult to manage and mitigate the declines of these fragile ecosystems. Coral reefs across the world have recently experienced the longest bleaching event on record (from 2014 to 2016) (Cressey 2016) and reports are now starting to highlight that many are experiencing bleaching for a further consecutive year (NOAA Coral Reef Watch 2017). However, with the use of hind cast modelling spatial variation in warming trends, thermal stress events and temperature variability has been mapped back as far as 1985 (Heron et al. 2016). This study suggested that over 97% of the reefs assessed had experienced positive sea surface temperature (SST) trends since 1985, with 60% experiencing significant warming. Furthermore, the frequency of thermal stress exceeding bleaching thresholds has increased three-fold between 1985-91 and 2006-12; a trend which climate model predictions suggest will continue (Heron et al. 2016). This has led some to suggest that, as early as 2054, we will likely see annual severe bleaching (ASB) episodes on a large proportion of the world's reefs. This is assuming emissions follow the Intergovernmental Panel on Climate Change's (IPCC) Representative Concentration Pathway (RCP) scaling of 4.5, whereby greenhouse gas emissions peak around 2040 and then decline. However, if climate conditions fall under the RCP8.5 scenario (i.e. emissions continue to rise throughout the 21st century) ASB conditions are predicted to occur 11 years earlier (i.e. by 2043). It should also be noted that there is likely to be significant spatial heterogeneity in these patterns, with reefs at different locations experiencing variable warming around the overall mean. For example, high-latitude reefs in Australia, Hawaii and India are predicted to have at least 25 years before they experience ASB conditions (under RCP4.5), whilst reefs nearer the equator are predicted to experience these conditions in less than 10 years (Heron et al. 2016). This has led some to propose that the Intended Nationally Determined Contributions (INDCs), which were submitted under the Paris Agreement as of April 2016, will do little to aid reefs in adapting or acclimatising prior to the occurrence of ASB events in the majority of locations (Heron et al. 2016).

In response to the rapid decline of many marine organisms, particularly those associated with coral reefs, there has been a marked rise in the designation of large scale marine protected areas (MPAs) (Ban et al. 2017). This has been coupled with an increase in research focusing on adaptability and response of corals to future climate-ocean scenarios (reviewed in Sweet and Brown 2016). However, decisions in locating MPAs are often based on data and requirements at the larger ecosystem scale

and not specifically focussed on coral reefs. There is substantial evidence that MPAs can improve the abundance and diversity of organisms in the higher trophic levels, such as fish communities associated with reefs (McClanahan et al. 2006). However, there is significant discussion over the effectiveness of MPAs on the conservation of corals and across the breadth of all reef organisms (Mouillot et al. 2016).

In contrast, reef restoration offers a more focussed conservation methodology. This has been used in a wide range of contexts, from areas which have been damaged due to ship groundings and hurricanes, to tourism driven projects around resorts and islands (Young et al. 2012). The majority of these projects have relied on taking fragments from parental colonies to utilise as the basis for new colonies i.e. asexual fragmentation (Young et al. 2012). Focus is now shifting to the development of methods using sexual reproduction as a more efficient and productive way to generate new colonies, making a larger number of transplants available which offers the potential for restoring reefs at larger spatial scales (Linden and Rinkevich 2017). However, with both methodologies, there are significant difficulties to overcome, such as project costs, a limited number of hosts in these degraded habitats, and the potential impact of founder effects associated with the utilisation of restricted numbers of parental colonies (Edwards et al. 2015).

More recently, less traditional methods of conservation have been proposed. Here we highlight two such methods which have generated considerable interest. The first is human assisted evolution (HAE), and the second is the use and manipulation of beneficial coral microbes (BCM). The concept around HAE in coral conservation developed from the observations that some corals have recently shown contrasting patterns of bleaching at the individual level within a reef, indicating the presence of a mechanism for natural adaptive response to thermal stress (Guest et al. 2012). However, it has been suggested that the natural rate of such adaptation may not be rapid enough in the face of the current rate of climate change (van Oppen et al. 2015). HAE aims to accelerate these naturally occurring evolutionary processes (e.g. random mutations, natural selection, acclimatization and changes in the microbial symbiont communities) (van Oppen et al. 2015). For example, fragments from only the more thermally robust colonies could be utilised for reef restoration projects, artificially enhancing the selection process (within the thermal context). In contrast, the use of BCM is aimed more specifically at enhancing the resistance of corals to the increasing devastation caused by disease outbreaks (Peixoto et al. 2017), taking a probiotic approach to aid corals in the face of thermal stress and increased risk of disease (van Oppen et al. 2015; Peixoto et al. 2017). This approach is analogous to a more established approach used in agriculture known as Plant Growth Promoting Rhizopheres (PGPR) (Dobbelaere et al. 2003), which has been used to directly or indirectly promote plant growth and development for many years, via the manipulation of plant root associated microbial communities.

Conservation biology in general is a crisis discipline (Soulé 1985) and coral conservation is becoming one of the most important arenas within the discipline. However, whilst we recognise both the urgency for coral conservation, and the large potential offered by both HAE and BCM, we advocate caution in utilising these methods at this time. For example, although there is substantial established literature on the partnership between reef building corals with symbiotic algae, we still do not understand enough about the ecophysiological responses of corals to climate change and the associated responses of the microbiome (i.e. the coral/host associated bacteria, viruses, fungi and protozoa) and the pathobiome (the microbial community linked with coral disease dynamics)

(reviewed in Sweet and Brown 2016; Sweet and Bulling 2017). Knowledge of all of these will be important in order to assess the risk of such mitigation measures appropriately. The concern is that we may use potentially powerful beneficial conservation tools to inadvertently inflict greater harm on reefs in the long term. For example, selecting for traits such as resistance to higher SST might lead to a genetic bottleneck reducing the capacity of corals to adapt to future changes in environmental conditions. There are also often other trade-offs associated with HAE which have been observed in other organisms such as plants e.g. a reduction in growth rates or a higher susceptibility to disease (Brown and Rant 2013).

In addition to changes which will be occurring on a genetic level, coral fragments obtained in the field, grown at a different location (even one close by), and then translocated back on the reefs may be being transplanted with new and potentially harmful microbial associates. Indeed, it has been well documented that the microbiome associated with corals can change rapidly (i.e. within a matter of hours), especially when corals are housed in aquaria (Kooperman et al. 2007; Ainsworth and Hoegh-Guldberg 2008). What we do not know, however, is how this change on a microbial level influences the coral 'holobiont' as a whole (i.e. on a physiological, genetic, metabolomic, and/or proteomic level). There is also the real possibility that we may introduce specific pathogens from aquarium settings to the wild if this practice is undertaken. For example, it has been hypothesised that the ciliates which have been proposed as secondary pathogens associated with the suite of 'white diseases' found in corals were transferred to the Caribbean from the Indo-Pacific, via ballast waters in ships and/or via the aquarium trade (Sweet and Séré 2016). Although conclusively identifying the original source of the ciliates is likely to be impossible, the finding that the same ciliate species (*Philaster lucinda*) has been found throughout the Caribbean and the Indo-Pacific with identical DNA sequence matches (Sweet and Séré 2016) provides strong circumstantial evidence for the recent transfer of the pathogen. There are other potentially harmful species associated with corals which are, to date, more commonly associated with aquariums, including the likes of the '*Acropora* Eating Flatworm' (*Amakusaplana acroporae*) and the 'Red Bug' (*Tegastes acroporanus*) (Sweet et al. 2011; Rawlinson and Stella 2012). We emphasize that here we are speculating on the potential for the spread or introduction of such parasites from aquariums or from locations used to grow new coral fragments. However, larger organisms have been released into new marine habitats with profound detrimental consequences. For example, two predatory Indo-Pacific lionfish (*Pterois volitans* and *P. miles*) have been introduced to the Caribbean and Western Atlantic (Côté et al. 2013). The rapid colonisation has led to negative ecological consequences such as a 95% reduction in the abundance of small fish at some sites. Population models have predicted that whilst culling can reduce lionfish abundance substantially, removal rates will need to be high and, despite the deployment of significant funding and effort, there has been little progress in mitigating the impacts (Côté et al. 2013). A particularly relevant example is the impact of the chytrid fungus *Batrachochytrium dendrobatidis* and its emergence as a severe threat to global amphibian populations. First described by Longcore et al. (1999), the fungus is causing a global pandemic linked to the decline of hundreds of amphibian species across the globe (Skerratt et al. 2007). Anthropogenic activities and commercial trade are strongly implicated in the spread of the disease through global movement of animals, the introduction of non-native infected individuals, and the spread of infection in captive populations and via water discharge (Fisher and Garner 2007; Kolby and Daszak 2016). The long term implications of the disease are not clear, but will go well beyond the loss of amphibian species. At the very least, coral translocations need to be complicit with the

IUCN guidelines (IUCN/SSC, 2013) in order to demonstrate feasibility, the assessment of risk and plans for long term monitoring and management.

Conservation of coral communities is a priority and requires a rapid response. However, decisions about appropriate conservation strategies must also factor in the risks of employing mitigation activities for which we lack substantial knowledge of the underlying mechanisms and dynamics. The proposed conservation methods of human assisted evolution and the use of beneficial coral microbes offer significant potential benefits in supporting coral conservation, particularly in regard to increasing the rate at which managed reefs could respond to the rapid rate of climate change. However, here we advocate caution in applying these methods too rapidly before the potential harmful impacts can be assessed to a greater degree. There will undoubtedly be many who will suggest that the rapid rate of coral decline, and the need to act urgently, will outweigh the potential risks in using HAE and BCM with our current state of knowledge. We do not necessarily disagree, but the potential risks of such actions need to be highlighted and discussed within the coral conservation community.

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