Corals as canaries in the coalmine: towards the incorporation of marine ecosystems into the ‘One Health’ concept

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Abstract

‘One World – One Health’ is a developing concept which aims to explicitly incorporate linkages between the environment and human society in order to facilitate optimising aspects of health. Historically, work has concentrated on aspects of disease, in particular on emerging zoonoses, with a focus on terrestrial systems. Here we argue that the marine environment is a crucial component of the ‘One World – One Health’ framework, and that coral reefs are the epitome of its underlying philosophy, providing vast contributions to a wide range of ecosystem services with strong and direct links to human society. Further, the sensitivity of corals to climate change, and the current emergence of a wide range of diseases, make coral reefs ideal systems for studying in order to develop our understanding of the links, impacts, and feedback mechanisms that can affect human and ecosystem health. There are well established protocols for monitoring corals and global networks of researchers, but there are substantial challenges in studying these ecosystems. We explore these challenges and conclude with a look at how developing technology offers ways of addressing them in the future. A greater integration of coral reef research into the ‘One World – One Health’ framework will enrich our understanding of the complex links within and between ecosystems and human society, supporting the development of measures for improving the health of both humans and the environment.

KEY WORDS; SCTLD, Vibrio, coral disease, health indices, reefs, zoonosis

Introduction

‘One Medicine’ was a concept originally proposed in the mid-19th century advocating the integration of human and veterinary medicine in responses to zoonoses. More recently, in 2004, the concept was broadened to explicitly include ecosystems within its framework. This came with a name change, ‘One World – One Health’ (Destoumieux-Garzón et al., 2018). ‘One Health’ is now defined by the Centers for Disease Control and Prevention (CDC) and the World Health Organisation (WHO) as *‘a collaborative, multisectoral, and transdisciplinary approach (working at the local, regional, national, and global levels), with the goal of achieving optimal health outcomes recognizing the interconnection between people, animals, plants, and their shared environment’* (<https://www.cdc.gov/onehealth/index.html>). The underlying premise of this concept is that the health and wellbeing of the human population will be increasingly difficult to maintain on a polluted planet suffering from extensive overexploitation, habitat degradation and global environmental change (Destoumieux-Garzón et al., 2018). The One Health concept therefore provides a broad and integrated framework for studying complex systems and mechanisms leading to undesirable effects such as disease emergence.

Nevertheless, to date, the vast majority of studies adhering to the One Health concept have focused on emerging zoonosis, particularly those originating in domestic animals and livestock (Day, 2011). In contrast, relatively few studies have addressed the importance of wildlife disease and ecosystem health for providing highly crucial services in the One Health context (Jenkins et al., 2015). This is not surprising as the successful integration of the environmental component hinges on the provision of a clear and concise definition of ecosystem health - a challenging task considering the complexity and interconnectedness of natural systems. Nevertheless, it is a task of central importance, which is reflected in the growing demand from administrators, managers and politicians for detailed information on the current and ideal health states of ecosystems. This demand is driven by the necessity for the use of ecosystem health assessments to guide decision-making and evaluate trade-offs between development, conservation and human health.

A clear example of the relevance of One Health is the recent outbreak of the novel coronavirus nCoV-2019. In the case of nCoV-2019, bush meat trade and consumption was the likely source of transmission to humans, a direct link between animal and human health. The virus is thought to have originated from horseshoe bats (*Rhinolophus affinis*) (Zhou et

al., 2020), with the possibility of Malayan pangolins (*Manis javanica*) as an intermediate host (Wong et al., 2020). This example reflects the classic view of ‘One Health’ (i.e. transmission of disease to humans and/or kept animals). However, in order to ensure sustainable and long-term human health, a wider perspective and the inclusion of feedback effects between humans and the environment need to be considered. Examples clearly highlighting the importance of such interactions include decreasing provision of ecosystem services driven by climate change (Zinsstag et al., 2018) and negative effects on wellbeing and mental health resulting from deteriorating environments (Dietz et al., 2009).

Looking towards our oceans

The past emphasis of One Health studies on human and animal disease has resulted in a primary focus on terrestrial environments. However, human wellbeing and health fundamentally depend on the oceans and seas that surround us. More than 40% of the global human population lives in areas within 200km of an ocean, and 12 of the 15 most populated ‘megacities’ are coastal (Jouffray et al., 2020). More than 3 billion people rely on seafood as an essential source of protein and nutrition, and fisheries and aquaculture directly employ 59.6 million (Friedman et al., 2020). Considering this heavy dependency on, and extensive human-wildlife interactions within the oceans, there has been surprisingly little attention given to zoonotic diseases originating from the marine biome. Those that have been described are usually associated with parasites such as trematodes, cestodes and nematodes (Ogawa, 1996), or bacterial infections, including *Mycobacterium marinum*, *Aeromonas hydrophila*, *Vibrio vulnificus*, *V. damsela*, *Edwardsiella tarda*, and *Streptococcus iniae* (Haenen et al., 2013). The importance of the connectivity within the oceans and the strong interface with human populations has been highlighted by recent work showing that seagrass meadows are able to reduce the relative abundance of bacterial pathogens capable of causing disease in humans by over 50% (Lamb et al., 2017). The same study also demonstrated that seagrass meadows offer the same protection to their neighbouring coral reef ecosystems and coral disease prevalence was 50% lower on reefs with adjacent seagrass meadows (Lamb et al., 2017).

Additionally, oceans-based research within the One Health framework has followed a very narrow definition of ‘marine ecosystem health’, tending to focus on aspects of disease and ignoring the wider aspects of ecosystem health. However, there is a wealth of evidence on the oceans’ role in global oxygen production, carbon sequestration (‘blue carbon’) and cost-effective natural buffers protecting coastal communities against flooding and coastal erosion

(Friedman et al., 2020). Despite our clear reliance on these ocean ecosystems, we continue to put them under immense stress. Global increases in sea temperatures, sea level rise and ocean acidification are having profound and well documented impacts. However, in addition to these global changes, many more regional drivers are much less appreciated, despite their large cumulative impact. For example, with increases in population, economic activity and urbanization, land reclamation has become common practice (Jouffray et al., 2020), often resulting in the cutting down of mangroves and the smothering of seagrass beds and reefs. Countries such as Dubai, China and the Maldives, are world leaders in these often large scale reclamation projects, extending their coastlines by hundreds of square kilometres every year (Jouffray et al., 2020). As we confront a future of increasing global mean surface temperatures, increasing environmental variability and increasing human populations, it has been anticipated that these stressors will be amplified and likely result in large negative feedback loops exacerbating negative impacts on human wellbeing (Friedman et al., 2020).

Canaries in the Coal Mine

Coral reefs cover less than 0.1% of the world's oceans but harbour upwards of 25% of all marine life (Spalding and Grenfell, 1997). The social, cultural and economic value of coral reefs is estimated at US\$1 trillion (Heron, 2018). However, they are under threat from a number of anthropogenic stressors including impacts associated with climate change, overfishing and pollution. Indeed recent projections indicate that climate-related loss of reef ecosystem services will total US\$500 billion per year or more by 2100, with the greatest impacts felt by people who rely on reefs for day to day subsistence (Heron, 2018). In 2008 a global assessment of reef health considered 19% of reefs degraded beyond repair, and identified 15-40% at severe risk of collapse (Wilkinson, n.d.). By 2011, more than 60% of reefs were classified as under immediate and direct threat from one or more local sources, a percentage which has since increased to 75% (Burke et al., 2011). Indeed, 21 out of 29 reef systems (72%) which are classified as World Heritage Sites have been exposed to severe and/or repeated heat stress in recent years (Heron, 2018).

The demise of the health of reefs, on a global scale, has primarily been attributed to the phenomenon known as coral bleaching (i.e. expulsion of the coral's symbiotic algae). Mass-bleaching events are now increasingly common and, for example, the Great Barrier Reef in Australia is currently experiencing the third such event in five years (Figure 1) [17, 18]. In 2016 and 2017, reefs on a global scale were exposed to an extended period of heat stress

(Figure 1), only the third such event since the start of records in the 1980s (Hughes et al., 2018). During this pan-tropical stress event, locations known for their high coral cover and diversity such as the Great Barrier Reef and the Maldives, were affected by large scale coral die-offs.

Bleaching represents a highly visible stress response (one which can be viewed from the air for example) and can be reasonably accurately predicted (for example NOAA's Reef Watch, which can be used to predict the likelihood of bleaching events on a global scale - Figure 2). Consequently, corals have been described as 'the canary in the coal mine' as they are reliable indicators of threats associated with environmental change and other local stressors such as pollution. After a bleaching episode, reefs lose less resistant coral species or heat-sensitive colonies and their associated biocenosis (Van Woesik et al., 2011). Such stress events not only affect alpha diversity at individual reefs, but also lower beta diversity (at the landscape level), decreasing the uniqueness of individual reefs. In terrestrial environments anthropogenic activities have been key drivers of ecosystem homogenisation (Carvalho et al., 2013a). In marine systems, climate warming is likely to have analogous effects. Only a small fraction of coral species are predicted to be able to withstand recurrent heat-stress, and these more resistant species are expected to increase in dominance across reefscapes (Kubicek et al., 2019). The associated shifts in relative abundance and species occurrence and level of homogenisation can be challenging to quantify (see Box 1). Nevertheless, functional and taxonomic homogenisation of coral communities are an essential aspect of increased stress levels, and impacts will propagate to associated fish and invertebrate communities resulting in substantial losses of overall genetic diversity.

Monitoring reef health

Large-scale global initiatives, aimed at monitoring coral reef health have already been put in place. One example is the Global Coral Reef Monitoring Network (GCRMN; <https://gcrmn.net/>). This incorporates "Essential Ocean Variables" within their framework and relies on the Driver Pressure State Impact Response (DPSIR) framework which was originally developed by UNESCO and UNEP. DPSIR integrates ecological research with social management and politics. However, for such monitoring networks to be effective, clear definitions and measures of ecosystem health are required. Such measures are usually instantiated in the form of health indices. For coral reefs, 'time-integrated bleaching thresholds' have been developed, based on *in situ* measured temperature data (Berkelmans,

2009). These were adopted after the first pan-tropical bleaching event in 1998 as an early warning system, and have been augmented by satellite-based early warning systems (i.e. “HotSpots”, or “ReefTemp” (Maynard et al., 2008)) which measure occurrence and magnitude of instantaneous heat stress (Figure 2). These are now commonly used to indicate the number of “Degree Heating Weeks” a reef is exposed to (i.e. how much heat stress has accumulated in an area over the past 12 weeks). Although such tools often correctly predict when reefs will bleach, anomalies frequently occur i.e. reefs exceed temperature thresholds without their corals bleaching (Berkelmans, 2009). Some have suggested that this higher resilience is a result of slow but measurable adaptive response to periodic heatwaves (Cooke et al., 2020). In other environments, turbidity has been proposed as a possible cause for some reefs escaping expected bleaching (Sully and van Woesik, 2020). However, both adaptation and the beneficial turbidity effects are still heavily debated and require further exploration (Duarte et al., 2020).

Another measure of reef health, the ‘bleaching and mortality index’ (BMI) was also developed after the 1998 mass bleaching event (McClanahan et al., 2004). The BMI categorises individual corals as either; (i) ‘unbleached’, (ii) ‘moderate’ (pale to 50% bleached), (iii) ‘severe’ (50–100% bleached) and (iv) ‘dead’. The weighted sum is then calculated and scaled from 0-100 according to the formula

$$\text{BMI} = (0 \text{ } c_i + 1 \text{ } c_{ii} + 2 \text{ } c_{iii} + 3 \text{ } c_{iv}) / 3 \quad (1)$$

where c_i to c_{iv} represent the percentage of corals in the four categories listed above.

Such indices have highlighted significant heterogeneity among coral species in their ability to cope with thermal stress (McClanahan et al., 2004). Differences in sensitivity and community composition can filter to the scale of reefs and cause differential sensitivity to climate change in varying ways. For example, corals from the genera’s *Acropora*, *Stylophora* and *Pocillopora* are often labelled as thermal sensitive, whilst those from *Cyphastrea*, *Goniopora*, *Galaxea* and *Pavona* are considered generally more resistant (McClanahan et al., 2004). Similar partitioning has been applied to life history strategies. Those identified as ‘competitive’ in nature grow quickly, shade out other genera and dominate communities in ideal non-stressed environments (e.g. Acroporids). ‘Stress tolerant’ corals are characterised by massive domed morphologies, high fecundity, as well as slow growth rates and are often more abundant in stressed environments (Darling et al., 2012; Grimsditch et al., 2017). A third group classed as ‘weedy’ are small and fast growing, but

have a high population turnover and often represent an early succession state after disturbance. Remaining corals are grouped together as ‘generalists’ and exhibit traits spanning the other categories (Darling et al., 2012; Grimsditch et al., 2017) .

Other reef ecosystem health indices aim to account for a greater taxonomic diversity of reef organisms in their evaluation procedures. Examples are the ‘reef health index’ (RHI) and the ‘coral health index’ (CHI) (See. www.healthyreefs.org, (Kaufman et al., 2011)). The RHI is relatively simple and combines four key general ecosystem indicators: live coral coverage, fleshy macroalgae coverage, biomass of herbivorous fish and biomass of commercially important fish. These indicators are averaged in order to obtain values on a range from 1 (“Critical”) to 5 (“Very Good”). In contrast, the CHI uses three sets of indicators: broad assessments of the benthos (including the coverage values of both encrusting coralline algae and live coral), total biomass of reef fish, and concentration of *Vibrio* spp. in the water column. These indicators are averaged in a similar manner to that of the RHI, but the scale runs from zero (very degraded) to one (very healthy).

To our knowledge, only one study has systematically compared the RHI against the CHI (Díaz-Pérez et al., 2016), and the study simplified the CHI by removing the bacterial element which means we cannot really assess the usefulness of the microbial element in this health index. However, the results of the comparative study (Díaz-Pérez et al., 2016) did highlight significant differences between the two indices. Interestingly, only the RHI showed a correlation between the health grades and the species and functional group richness of fish at the scale of sites, and with the species and functional group richness’s and the Shannon diversity index of the fish assemblages at the scale of zones. Surprisingly, neither of the health indices were correlated with the metrics analysed for the coral diversity (Díaz-Pérez et al., 2016), possibly indicating a limitation of these indices. An important next step for the improvement of composite measures could be the introduction of reference sites (Box 1). Many freshwater indices (e.g. the European Water Framework Directive) do not apply absolute measures of ecosystem measures. Instead, they account for differences across regions and habitats (e.g. in a coral context, depth and latitude would be crucial factors) by comparing sites with a pristine reference site that shares key environmental characteristic with a target location. Such standardisation of indices would be especially important for super-regional monitoring and global assessments of coral health.

Coral disease and reef health.

223 The health indices mentioned above are strongly reliant on measures such as coral diversity
224 or coral cover. However, neither the RHI or the CHI directly account for (or record) coral
225 bleaching or diseased corals at the time of surveying. This is despite, bleaching and the
226 frequency of occurrence of opportunistic diseases being excellent indicators of stress at the
227 community level due to climate-mediated, physiological stresses which frequently
228 compromise host resistance.

229 In 2009, a landmark study highlighted the additive effects of a mass bleaching event and a
230 subsequent outbreak of a disease in the Virgin Islands (Miller et al., 2009). Miller and
231 colleagues found a 13-fold increase in mortality when disease and bleaching co-occurred,
232 compared with when only bleaching was observed. A more recent study, by Brodnicke et al.
233 (2019) showed similar trends, this time on the GBR, Australia. They found that, during the
234 2016-17 mass-bleaching event, monitored corals which suffered from disease as well as
235 bleaching were seven times more likely to die than if the coral was only bleached (Brodnicke
236 et al., 2019).

237 However, even in the absence of major acute disturbances (i.e. bleaching), corals can be hit
238 by small-scale disease outbreaks, which may cause relatively high rates of mortality in their
239 own right. For example, in 2008 at Christmas Island, 36% of Acroporids surveyed suffered
240 total mortality from an outbreak of disease, causing total coral cover to decline from the
241 already low coral cover of 7.0% to 0.8% over an 8 month period (Hobbs et al., 2015).
242 Throughout the Caribbean another disease, known as ‘stony coral tissue loss disease’
243 (SCTLD) is currently spreading (Box 2). SCTLD was first observed in 2014 in Southeast
244 Florida (near Miami) and has been reported to infect 29 species (recorded to date). While
245 SCTLD seems to be largely temperature independent, it nevertheless represents a major
246 disturbance of ecosystem health and many scientists, conservationists and ecosystem
247 managers are rightly concerned over its potentially long-lasting effects (see Box 2 for more
248 detail on SCTLD).

249 Diseases are a normal part of the natural dynamics and turnover in coral populations and
250 communities. However, the observed increase in prevalence and severity on reefs will likely
251 undermine the capacity for these ecosystems to recover from other disturbances (annual
252 background mortality rates can generally vary from 1 to 30% (Pisapia and Pratchett, 2014)).
253 The Caribbean is already regarded as a degraded coral reef region and exemplifies the multi-
254 causality aspects of reef loss. Caribbean corals have suffered from previous coral disease

outbreaks (white band disease affecting the historic reef builders *Acropora palmata* and *Acropora cervicornis*) (Cramer et al., 2020), overfishing, high levels of pollution, land reclamation, the die off of the urchin *Diadema antillarum* in the 1980s unbalancing ecological dynamics, the impacts of more frequent and strong tropical storms, not to mention climate change (Edmunds, 2019).

In many ecosystems, understanding health means understanding the primary causes of diseases as outbreaks often co-occur alongside multiple stressors. However, the causal agent(s) for many coral diseases is unknown. Out of the 28 diseases described for corals worldwide, only 6 have had pathogens assigned to them (Sheridan et al., 2013), despite over 40 years of research in this field. The ‘known’ pathogens include *Vibrio coralliilyticus*, described as the most well characterised coral pathogen and associated with coral bleaching and white syndromes (Pollock et al., 2010; Ushijima et al., 2014) (see Box 3), *Serratia marcescens* (cause of white pox disease – WPX also known as *Acropora* Serratiois - (Sutherland et al., 2015)), *Pseudoalteromonas pirati* (cause of *Montipora* white syndrome – WS (Beurmann et al., 2017)), *Vibrio tubiashii* (cause of *Porites* white patch syndrome – PWPS (Séré et al., 2015)), *Vibrio shiloi* (cause of bleaching in the coral *Oculina patagonica* (Kushmaro et al., 2001)) and *Thalassotalea loyana* (formally *Thalassomonas loyana* & cause of white plague disease – WPD) (Thompson et al., 2006).

Nonetheless, even for these ‘well described coral pathogens’ there remain some inconsistencies and conflicting results among various studies (see Box 3). For example, *Montipora* WS can be induced in controlled laboratory conditions by two different agents - either *V. coralliilyticus* or *P. pirati* (Beurmann et al., 2017; Ushijima et al., 2014) leading to critical questions about whether these are indeed the pathogens of the wild disease? Other coral diseases such as *Acropora* Serratiois appears to have shifting aetiologies (Kathryn P. Sutherland et al., 2016). *S. marcescens* was originally established as the causal agent in 1994 (Kathryn P Sutherland et al., 2016), and was shown to be transmitted via untreated human sewage (Sutherland et al., 2010). Later, studies documented diseases bearing striking similarities to *Acropora* Serratiois on reefs without such waste water contamination and, despite considerable screening efforts, *S. marcescens* was never found (Lesser and Jarett, 2014).

Regardless of the uncertainties associated with some key fundamental ecological questions around the true causal agents of many of the described coral diseases, it is clear that

outbreaks of a disease can have major impacts on the reef ecosystem and strongly interact with mass bleaching events. Therefore, we argue that disease prevalence should be a central aspect of any ecosystem assessment evaluating the health of reef communities.

Methodological advances and challenges in monitoring coral reef health

Corals present many challenges for researchers monitoring their abundances, health and disease states. Their general plasticity in physiology and growth forms, as well as difficulties in ascertaining the reason behind tissue loss, complicate community assessments. When surveying healthy corals, assigning the correct taxonomy can be a difficult task. For example, phylogenetically distinct individuals may have visually very similar structures and morphology. Growth patterns often lead to non-regular shapes and edges and intricate forms. Therefore, examination of individual colonies at different magnification scales will often be required in order to accurately identify species, and help to ensure the counting of all corals on any given survey.

Classic survey methods to document coral diversity (alpha and beta; Box 1), coral cover and coral health state include line intercept transects (LIT), belt transects, point intercept transects (PIT), and quadrats or tow boards (manta tows) (Facon et al., 2016; Jokiel et al., 2015). The most appropriate method to use will be context dependent, but comparisons of the methods suggest that they are all reliable in providing a good first estimate of coral cover, diversity and/or health (Facon et al., 2016; Jokiel et al., 2015). Inconsistencies between methods can however emerge when more detailed data is required such as the total number of coral colonies present and their health states (Facon et al., 2016; Jokiel et al., 2015). For example, identifying disease signs can be difficult if attempted by non-experts. Arabian Yellow Band Disease (AYBD) has been featured in a number of papers (Johan et al., 2017). However, in some cases, encrusting sponges (i.e. not a disease) may have been responsible for the yellow colouration on the corals (Figure 7.8e for example in (Riegl and Purkis, 2012)). Rapid surveys, or surveys conducted by researchers unfamiliar with diseases in corals, may therefore lead to false positive results for disease presence. The same issues arise when describing the ‘white diseases’ in corals i.e. a lesion characterised by a sharp demarcation between apparently healthy tissue and the denuded coral skeleton – not unsimilar to the field signs of SCTLD (Box 2). In the majority of disease surveys such ‘white diseases’ are usually referred to as ‘white syndrome’ (WS) in the Indo-Pacific and ‘white band disease’ (WBD) or white plaque disease (WPD) in the Caribbean. However, it is likely that other causes of

similar signs on coral may be mistaken as a disease. For example, the feeding scars of coralivores (for example the snail *Drupella cornus*, sea stars such as the Crown of Thorns *Acanthaster planci* or their smaller cousins the Cushion stars *Culcita novaeguineae*) are often mistaken as a ‘white syndrome-like’ disease and therefore classified in disease surveys in error [Sweet pers obs, 52]. However, we recommend that such signs, if reliably identified could benefit from being classified as ‘non disease’ during any survey. Indeed, many of the invertebrates feeding on corals have been credited as vectors of disease (e.g. the snail *Coralliophila abbreviate* for WPX, WBD, and WPD (Shore and Caldwell, 2019)) and may drive disease propagation.

As we move into an era with an increasing need to monitor ecosystem health, technological innovations may enable us to gather more information and detail and, in some instances, improve our ability to sample larger areas relatively rapidly. Examples of these newer monitoring methods include 3D mapping of reefs (Zawada et al., 2019), blue light excitation of photoquadrats - using florescence imaging systems such as FlourIS (Zweifler et al., 2017), hyperspectral imaging (Teague et al., 2019), and light detection and ranging systems (LIDAR) (Sasano et al., 2016), infrared sensitive surveillance technology cameras with high power IR-LEDs (Dirnwoeber et al., 2012), monitoring using artificial intelligence (González-Rivero et al., 2020), and the use of LANDSAT 8 OLI-TIRS combined with Sentinel-2A satellite imagery (Nurdin et al., 2019). The utilisation of machines, such as drones, underwater remotely operated vehicles (ROVs), and/or autonomous underwater vehicles (AUVs) for data collection, is also becoming increasingly frequent (Madin et al., 2019). Combining some of these developments with machine learning techniques is starting to show promising results for semi-automatically evaluating health states of corals on reefs. For example, the pairing of *in-situ* measurements and medium resolution imagery is making progress towards the development of a digital health chart for reefs, although the current scope only separates coral into broad health states (Nurdin et al., 2019).

Whilst digital imagery has greatly improved the speed of underwater data collection, the analysis of these images remains a bottleneck in reporting the observations. A recent study provides a promising framework for combining artificial intelligence (such as deep learning convolutional neural networks) and a high resolution monitoring method which may help to address this issue (González-Rivero et al., 2020). Indeed, this study found unbiased and high agreement between expert observations and automated artificial intelligence-based processing (97% similarity). Further, repeated surveys together with a comparison against existing

monitoring programs also showed that automated estimation of benthic composition was equally robust in detecting ecological transitions (e.g. changes in health state) (González-Rivero et al., 2020). Such results suggest that the use of new technology can offer continuity with existing monitoring data (González-Rivero et al., 2020). Using such approaches, estimates point to an increase by x200 with regard to the speed of reporting and analysis at a fraction of the cost (1%) (González-Rivero et al., 2020). However, although it is acknowledged that such systems offer substantial potential, general applicability needs to be tested carefully in new environments and under changing ecological conditions. Most likely, different sampling techniques should be considered as complementary, especially as new exciting possibilities to combine data from different sampling methods show steady positive trajectories in their development (Miller et al., 2019).

Future directions and monitoring needs

Besides imaging techniques, there are a number of other emerging possibilities to enhance the available data to monitor reef health. For example, the use of citizen scientists for reef monitoring is a very interesting and promising option/tool, especially in low-income countries with limited conservation budgets (Stelfox et al., 2020). For example, citizen science approaches have been beneficial when creating biodiversity indices (Bourjon et al., 2018). Further, programs such as Reef Check were developed to carry out snapshot global assessments of reef health using just such citizen science approaches (Hodgson, 2001). However, due to the required levels of expertise to reliably identify disease states (outlined in this review), this may not be a solution for obtaining robust high-resolution reef health data. Even for the collection of basic data, citizen science projects need to overcome a number of challenges including risks of inconsistencies and data biases. However, if these are mitigated through clarity in protocols and appropriate levels of training and validation, the development of open platforms (e.g. MERMAID , an online-offline app for coral reef data collection, https://collect.datamermaid.org/docs/MERMAID%20User%20Manual_Beta_updated%20July%202019.pdf) can provide additional, valuable data sources.

The use of environmental DNA analyses to monitor coral and fish communities is also showing great promise. However, due to the nature of the collected data, its application is likely to be limited to investigate diversity indices or abundances of specific taxa (Leduc et al., 2019). Recently a high-resolution, multi-trophic biomonitoring tool (targeting a number of key genes; CO1, 16S and 18S rRNA) highlighted the ability to detect a wide range of bony fish and elasmobranchs (244 taxa), crustaceans (88), molluscs (37) and echinoderms (7)

(West et al., 2020). However, there is currently contrasting evidence with regard to the reliability of such surveys. Some acknowledge difficulties with regard to identifying past the phylum level (Nguyen et al., 2019), whilst others illustrate capability of differentiating among coral taxa (Nichols and Marko, 2019). Regardless, with the advent of new ecogenomic sensors (e.g. the Environmental Sample Processor developed by Dartmouth Ocean Technologies Inc. and Sieben Laboratory Dalhousie University), eDNA will almost certainly be utilised more routinely as these molecular analytical techniques can be conducted subsea (DNA probes, protein arrays and qPCR conducted *in situ*) (<https://www.mbari.org/technology/emerging-current-tools/instruments/environmental-sample-processor-esp/>).

A number of scientists have also proposed the adaptation of other genomic tools to aid in the conservation and/or restoration of reefs on various scales. For example, the development and use of specialised biomarkers (Parkinson et al., 2019). Biomarkers are easily measured indicators of biological processes that can be utilised to predict or diagnose health, resilience or other key metrics. Typical biomarkers would be the presence or absence of known coral pathogens and, to this end, a monoclonal (dipstick) assay has been developed for the detection of *V. coralliilyticus* (Gharaibeh et al., 2013) (see Box 3).

Additionally, monitoring of host gene expression may function as a biomarker for targeting coral responses to stressors. Corals might upregulate or downregulate the expression of specific genes during perturbations and stress events (Gibbin et al., 2018). For example, two, PdC-lectin and concanavalin A-like transcripts become increasingly expressed after inoculation with *V. coralliilyticus* early in the disease process (day 6 to 12), then plateau on day 15 and return to control levels by day 18 (Vidal-Dupiol et al., 2011). The monitoring of activity in these genes has the potential to contribute to early warning systems as well as to provide insights into the physiological processes underlying responses to disease.

However, due to the current substantial lack of detailed knowledge about causes and mechanisms underlying specific diseases and the response of the host (see Box 3), the need for more holistic monitoring approaches has been emphasised. Instead of only focusing on the health state of the coral host for example, attempts to measure and map shifts in the community profiles of the corals microbial associates have substantially increased in frequency. Some studies have found that corals in favourable habitats host highly stable microbial communities or ‘microbiomes’, whilst those in unfavourable habitats host less

structured and more diverse communities (Roder et al., 2015). Furthermore, environmental perturbations have been shown to sometimes reduce the capacity of the host and/or the microbiome to regulate community composition, resulting in unstable microbial communities (Zaneveld et al., 2017). However, for an example showing contrasting results see (Sweet et al., 2019), which highlights the importance of not generalising across species and disturbance effects (Sweet et al., 2019). That said, the measure of associated microbial communities in corals, especially over extended temporal scales appears to offer highly rewarding insights into the health state of corals.

An interesting development of using microbial communities as biomarkers was a study that explored the use of microbial communities in reef seawater as an effective biomarker to monitor coral ecosystem stress (Glasl et al., 2019). In their study, microbes in the water had a higher diagnostic value than monitoring the microbial communities of the sediment or host-associated microbiomes (coral, sponge and macroalgae). However, the authors noted that the lack of changes seen in their host-associated microbiomes might have been due to the sensitivity (or lack thereof), offered from the relative simple sequence profiling technology utilised. Indeed, metaproteomic studies on reef sponges indicate that, whilst the microbial community composition can appear fairly stable with increasing water temperatures, disruption to nutritional interdependence and molecular interactions occurs prior to detectable changes in the community structure (Fan et al., 2013). Therefore, when considering the importance of microbes to reef health (and therefore their utility as a monitoring tool), more sensitive transcriptomic/proteomic or metabolomic approaches may be warranted in order to ensure reliable detection of microbial responses to environmental perturbations (Glasl et al., 2019).

Monitoring the corals' symbiotic algae may also offer a further route for assessing coral health and/or supporting a system for detecting the early onset of stress or disease. As mentioned earlier, coral bleaching is easy to see and monitor. Studies have also shown that a finer resolution can be achieved when corrected fluorescence of the host is measured (Santos and Shaw, 2019; Wooldridge, 2013). Water temperature increases often lead to decreased fluorescence of the host before the onset of visual bleaching signs (Santos and Shaw, 2019; Wooldridge, 2013). Such changes in functionality can be coupled with an initial increase in the algae density until they reach a thermal limit, and then rapidly decrease due to expulsion and then the characteristic visual bleaching signs ensue (Santos and Shaw, 2019; Wooldridge, 2013).

Finally, a potential future management tool is the identification of individual colonies with either disease resistance genotypes, differential disease susceptibility and/or differential infectiousness (e.g. super spreaders). Such an ability to categorise individual coral colonies would create the potential for targeted removal of highly infectious and/or susceptible corals (e.g. culling or lesion treatment). Such a management strategy could reduce transmission rates during disease outbreaks. Although this may seem impractical on a healthy reef with >40% coral cover, as reefs continue to degrade and reefs have reduced coral cover (e.g. reefs in the Caribbean), such a management strategy becomes a more feasible option. Indeed, with regard to the outbreak associated with SCTLD (Box 2), reef managers and governments are exploring the possibility of implementing such management options (Sweet pers obs). Understanding early signs of disease infection, like the monitoring of infection clusters (Sweet et al., 2019) or more simply catching physical disease signs early (see Box 2) may start to be utilised in the near future. Failure to react to early signs may result in the spread of diseases to possible unmanageable levels. Indeed, early warning of SCTLD was given (Precht, 2019), but no action was taken. What is becoming increasingly clear, is that any strategies undertaken to mitigate and prevent coral disease outbreaks in the future will almost certainly require some degree of ‘customisation’ i.e. a specific action for individual colonies, individual reefs or different geographically distinct habitats (see (Peixoto et al., 2019)).

Conclusions

The One Health concept acknowledges the interconnectedness of human societies and ecosystems, and recognises that the importance of these connections for human health is only going to grow as human populations and the extent of globalization increase. Currently the One Health framework is in a relatively early stage of development, having focussed primarily on zoonoses associated with domestic and livestock animals. The framework now needs to be developed further to include wider aspects of health, such as ecosystem functioning, impacts of climate change and consequences for mental health. Additionally, to date, there has been a strong terrestrial focus in One Health based research. However, the importance of the oceans for human society is very substantial. In particular, reef systems, which support high proportions of global biodiversity and are at the physical interface with humans, as well as the terrestrial-ocean interface.

In this paper we explored the One Health framework in a marine context and argued that coral reefs present excellent candidates to monitor ocean health and degradation. Coral reefs

are strongly linked to key ecosystem service provisioning, and directly interface with much of the world's human population, and are at the forefront of ecosystem impacts from climate change. They show clear signs of stress, primarily in the form of bleaching, and are suffering from increasing numbers, and prevalence, of diseases, indicating a severe perturbation of sensitive ecosystem dynamics. The canaries are singing, and we need to act.

However, despite their high potential for use in monitoring ocean health, corals present many significant challenges due to the complexity of their biology as well as of the associated higher-level ecological and socio-ecological interactions. However, the coral research community has already established standard survey methods and indices to quantify reef health and the impacts of diseases, and there are networks that feed into larger programmes linked to global human-environment monitoring initiatives led by UNESCO and UNEP. Additionally, the rapid technological development of monitoring tools provides a strong foundation for substantial progress in the near future. We therefore argue that the integration of coral reef research into the One Health framework will help to support the broadening of the framework and is an essential step in acknowledging the importance of marine ecosystem health for human health and wellbeing.

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891

■ Most severe bleaching
 ■ No or negligible bleaching

Figure 1. Areas of the Great Barrier Reef, Australia that showed severe forms of bleaching between 2016 and 2020. Surveys were conducted by researchers from the Australian Research Councils Centre of Excellence for Coral Reef Studies. Compared to the four previous bleaching events, there are fewer unbleached or lightly bleached reefs in 2020 than in 1998, 2002 and 2017, but more than in 2016. Similarly, the proportion of ‘most severe bleaching’ (red) in 2020 is exceeded only by 2016. In 2016, approximately half of the shallow water corals died on the northern region of the Great Barrier Reef, a result linked directly to bleaching and/or disease after the thermal stress – see (Brodnicke et al., 2019). By these metrics, 2020 appears to be the second-worst mass bleaching event of the five experienced by the Great Barrier Reef since 1998. The reefs showing no or negligible bleaching (green) in 2020 are predominantly offshore, mostly close to the edge of the continental shelf in the northern and southern GBR. However, offshore reefs in the central region were severely bleached in 2020 as they were in 2017 and 2016. Coastal reefs appear severely bleached at almost all locations, stretching from the Torres Strait in the north to the southern boundary of the Great Barrier Reef Marine Park in locations like Heron Island for example. These southern reefs had escaped bleaching in 2016 and 2017 - <https://theconversation.com/we-just-spent-two-weeks-surveying-the-great-barrier-reef-what-we-saw-was-an-utter-tragedy-135197>.

Figure 2. The National Oceanic and Atmospheric Administration's (NOAA) Coral Reef Watch (CRW) twice-weekly global 50km satellite Bleaching Alert Area outlines the areas where coral bleaching heat stress reaches various bleaching stress levels (a) Pacific and (b) Indian Ocean and Atlantic, based on their satellite sea surface temperature (SST) monitoring. The coral bleaching heat stress levels are defined as; 'No stress' = no bleaching, 'watch' and 'warning' = possible bleaching, 'Alert level 1' bleaching likely, 'Alert level 2' mortality likely. Products, (c) highlights the predictive capabilities of this tool with regard to threat of coral bleaching in the Maldives. These stills were taken on the 24th Feb 2020 for a and b and the 27th for c. It is predicted that the Maldives will see bleaching by the end of March, early April and reach Alert level 2 end of April, May if the current trends in increasing sea surface temperatures in this region continue.

Box 1: Habitat homogenisation in marine environments

In terrestrial environments habitat homogenisation has been identified as a major threat to global diversity (Carvalho et al., 2013b). In many instances, human activities do not necessarily reduce species richness at local spatial scales (Vellend et al., 2013). However, at a landscape level, habitat heterogeneity is degraded, which can lead to a reduction of beta-diversity and the loss of many rare and endangered species (Blowes et al., 2019; Britton et al., 2009).

In marine environments, the impacts of habitat homogenisation are comparatively underexplored. Nonetheless, future environmental pressures such as climate change have the potential to substantially alter not only alpha diversity but also exacerbate the uniqueness of individual reefs. Many reefs (in the Indo-Pacific) host more than 200 species of corals. This great diversity, and the associated structural complexity of coral reefs, support around 25% of total marine taxa richness (Burke et al., 2011). Climate change and the concomitant heat stresses for coral reefs are leading to enormous losses, but it is unlikely that all reefs will disappear. A more likely scenario is the survival of a small subset of more heat resistant species, leading to a loss of alpha diversity with associated functional and taxonomic homogenisation across reefs.

Despite its importance, habitat homogenisation is currently difficult to quantify if no detailed temporal data on coral diversity is available. Measurements of homogenisation usually rely on beta-diversity indices, which are calculated based on pairwise comparisons of two communities. However, for understanding the connection between heat stress and homogenisation, this leads to confounding effects as the communities usually differ in the levels of stress that they are exposed to. Consequently, it is difficult to relate beta-diversity and stress values to each other.

We want to introduce here the ecosystem homogenisation index (EHI) - a possibility to account for such affects and evaluate habitat homogenisation across environmental gradients. The EHI allows the computation of a beta-diversity value for an individual community i based on a weighted mean of all pairwise community comparisons n that involve the given community:

$$EHI_i = \frac{\sum_{k=1}^n S_{i,k \neq i} (1 - D_{i,k \neq i})}{\sum_{k=1}^n (1 - D_{i,k \neq i})}$$

where $S_{i,k}$ represents the community similarity score between site i and site k , and $D_{i,k}$ is the difference between the range transformed stressor levels of sites i and k . If multiple variables define stress levels (e.g. max temperature and days above 30°C per year), then a stress dissimilarity matrix can be used to compute D (see below). By using D for weighting pairwise community comparisons, EHI indicates the similarity to communities with similar stress level. Therefore, if a stressor leads to community homogenisation, a positive relationship between EHI and stress level will emerge.

Box 2 Coral Disease Case study 1: Stony Coral Tissue Loss Disease (SCTLD)

SCTLD is being heralded as unique among other coral diseases in that signs and dynamics of infections vary within and among affected corals. Different species display different rates of tissue loss (acute and subacute), lesion morphology (adjacent bleached zone or not) and lesion occurrence (focal and multi-focal). Here we illustrate a *Colpophyllia natans* colony from the U.S. Virgin Islands showing signs of SCTLD. Tissue loss for this colony averaged 293 cm²/day, and the entire colony (1m x 2m) died in just over 1 month after the first visual signs of the disease.

As is the case with many other coral diseases, we currently know very little about the aetiology of this disease, although it does appear to be bacterial in origin due to success with treating diseased fragments with antibiotics (Aeby et al., 2019; Meyer et al., 2019). The agent or agents responsible appear water borne, likely carried by ocean currents, but there seems to be no consistent pattern with SCTLD related to water temperature (Aeby et al., 2019; Meyer et al., 2019). Transmission of the etiological agents may be facilitated by phytoplankton and zooplankton blooms (associated with terrestrial storm runoff and eutrophication) as suggested for another coral disease, the acute *Montipora* white syndrome (Aeby et al., 2016). The rapid regional spread of SCTLD is being accredited to transmission of the pathogens via ballast waters of ships (Marine Safety Information Bulletin Ballast Water Best Management Practices to Reduce the Likelihood of Transporting Pathogens That May Spread Stony Coral Tissue Loss Disease, 2019).

At the time of writing, five possible causal agents have been proposed including an unclassified genus belonging to Flavobacteriales, a *Fusibacter* (Clostridiales), a *Planktotalea* (Rhodobacterales), a *Algicola* (Alteromonadales) and a *Vibrio* (Vibrionales) (Meyer et al., 2019). Some of these taxa also have been proposed as 'pathogens' in other studies. For example, *Fusibacter* as the cause of the white plague diseases (WPD) in the Pacific (Roder et al., 2014), and *Planktotalea* and the *Vibrio* for WPD in the Gulf of Mexico (Sunagawa et al., 2009). However, it has also been suggested that SCTLD may be a number of separate diseases. Indeed, (Aeby et al., 2019) indicated that during manipulative experiments, different species of coral and different colonies of the same species exhibited either acute or subacute lesions, suffering varying levels of infection or end response. Interestingly, corals either succumbed quickly to the disease or lost signs of the initial disease, with recovery being either complete or leading to a fatal re-infection months later. These differential disease dynamics are accompanied by variation in types and levels of defence against the onset of disease among coral families, genera, species and even individuals (Baums et al., 2013; Granados-Cifuentes et al., 2013; Muller and Van Woesik, 2014). Embracing this complexity is a fundamental challenge for assessing ecosystem health of coral reefs. At the same time, the sheer scale of impact SCTLD and other diseases can have on reefs also emphasises the importance of integrating coral disease in such assessments.

Box 3 Coral Disease Case Study 2: *Vibrio coralliilyticus*, or know thy enemy

The complexity of coral pathogens and difficulties in understanding coral-disease dynamics are well outlined by the case of *V. coralliilyticus*. Under normal conditions, the relationship between corals and *V. coralliilyticus* is thought to be commensal (Gibbin et al., 2019). For example, *V. coralliilyticus* bacteria have been found in healthy *Porites compressa* and *Montipora capitata* (Shore and Caldwell, 2019). However, when the coral becomes stressed, bleaching can be initiated by "inoculations" of *V. coralliilyticus*.

When a coral is inoculated with *V. coralliilyticus*, the polyps initially retract into their calices, which may be an attempt by the coral to isolate from the pathogens in the surrounding water. As with numerous other marine bacteria, *V. coralliilyticus* is believed to navigate microenvironments, and possibly locate new hosts, through chemotaxis (Ushijima and Häse, 2018). Once the first pathogens find the host, corals often spew bacterial laden mucus (Gibbin et al., 2019), thought to be a first line of defence. However, coral mucus and components within, such as dimethylsulfoniopropionate (DMSP), are chemo-attractants to *Vibrios* in their own right (Garren et al., 2014), suggesting that more *Vibrios* will be attracted to the health-compromised host. Pathogens within the coral are usually found in the oral epidermis (Gibbin et al., 2018). After entry into the coral, bacteria have been visualized inside the ectodermal cells (9 -13 days after initial inoculation) (Vidal-Dupiol et al., 2011). Here they form aggregates which appear disorganized and with the cellular organelles mostly lysed (Vidal-Dupiol et al., 2011). Interestingly, electron dense intracellular vesicles have also been observed to enclose *V. coralliilyticus* cells once they have entered the host, and these are thought to be a part of the coral's immune response (Gibbin et al., 2018).

Recent studies have also shown that *V. coralliilyticus*, similar to their cousin *V. cholerae*, carry the type VI secretion system (T6SS). T6SS is a nanomachine, capable of killing eukaryotic and bacterial prey by directly injecting toxic effector proteins into them (Guillemette et al., 2020). It is therefore hypothesised that *Vibrios* (once inside the coral) displace the host associated commensals causing community phase separation and/or increasing intra-guild predation. Indeed, early studies (Rozenblat and Rosenberg, 2004; Sussman et al., 2009) highlighted that *Vibrios* would need to kill the Symbiodiniaceae, which would result in bleaching. Given that the *V. cholerae* T6SS has been shown to behave in a similar manner with, for example, the amoeba *Dictyostelium discoideum* (Pukatzki et al., 2006), there is supporting evidence for these theories.

However, even though some of the key mechanisms of action between *V. coralliilyticus* and corals are relatively well researched and a theoretical framework of the processes is developing, there are still considerable gaps in our understanding as well as some observations which do not support the current views. The majority of inoculation experiments focussing on *V. coralliilyticus* have found that the number of cells needed to induce bleaching and/or tissue loss is between 10^6 to 10^8 cells ml⁻¹ (Gavish et al., 2018; Ushijima et al., 2016, 2014). However, this is magnitudes greater than one would expect for free swimming bacteria in the ocean (Gavish et al., 2018). To explain this aspect, fomites or vectors have been identified, which concentrate the bacteria, and this mode of transmission has been heralded as being vital for the progression of disease in many corals. Indeed, a number of other reef-associated organisms are known to harbour the *Vibrio*, including the blue spotted goby (*Asterropteryx semipuntata*), the oval butterflyfish (*Chaetodon lunulatus*), hinge-beak shrimp (*Cinetorhynchus* sp.) and feather duster worms (*Sabellastarte spectabilis*) (Shore and Caldwell, 2019). Additionally, ingestion of plankton, which are themselves infected by *V. coralliilyticus* (Ushijima and Häse, 2018), has also been hypothesised as another possible route of transmission. However, clear evidence directly linking them as vectors of the disease is lacking.

Further, and perhaps a more important consideration, is that only relatively few bacteria have ever been imaged within coral tissues (experimentally or in the wild), and then only during the various controlled inoculum experiments mentioned above. For example, in the study by (Gibbin et al., 2018), only 9 and 14 cells were identified in the coral tissues despite using high resolution imaging techniques (NanoSIMS coupled with microfluidics).

Additionally, not all strains of *V. coralliilyticus* appear pathogenic, and those that do, do not infect all corals in the same manner. For example, strain OCN014 appears virulent to *Acropora cytherea* but is unable to infect *Montipora capitata* (which is susceptible to infection by strain OCN008) (Ushijima et al., 2016). Part of this puzzle may be explained when looking at the behaviour of *V. coralliilyticus*' human-infecting cousin, *V. cholerae*. Both bacteria require the acquisition of key virulence factors to become toxigenic (Weynberg et al., 2015). For *V. cholerae*, there is a greater understanding of the mechanism. TCP (toxin co-regulated pilus) and the exotoxin cholera toxin (CTX) are both introduced via horizontal gene transfer (HGT), and are then encoded by a pathogenicity island and the genome of the single stranded DNA lysogenic filamentous bacteriophage (or prophage) CTXφ, respectively. The *V. coralliilyticus* isolate BAA450 (which induces bleaching at temperatures between 24-26°C, but not below) has a prophage genome that shares similarities to the genome of CTXφ. Interestingly, another strain (OCN008), which has been indicated as the aetiological agent in white syndrome in *M. capitata* (Ushijima et al., 2014) and appears to not be temperature dependant, is missing CTXφ, but has a prophage genome similar to another prophage also found in *V. cholera* i.e. K138 (Weynberg et al., 2015). These different prophages may be the reason why the different *V. coralliilyticus* strains have varying dependence on temperature before they are able to infect. Examining the impacts of mutants of pathogens has proved to be informative in helping to piece together these complex dynamics. For example, the outer member protein OmpU appears essential for infection of *M. capitata*, but mutants lacking the type IV MSHA pili are unable to effectively infect the coral. This has been hypothesised to be a mode of avoidance from the pathogen, in order for it to overcome additional host defences such as the muco-ciliary action at the coral surface.