A contribution to Mozambique's biodiversity offsetting scheme: Framework to assess the ecological condition of coral reefs

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Abbreviations and definitions

AGRRA	Atlantic and Gulf Rapid Reef Assessment protocol
AvCSA	Average colony surface area
BACI	Before-After-Control-Impact
BIOFUND	Foundation for the Conservation of Biodiversity
CHI	Coral health Index
COMBO	Conservation, Impact Mitigation and Biodiversity Offsets project
COTS	Crown of thorns starfish (*Acanthaster spp.*)
CPUE	Catch per unit effort
CRAMP	Coral Reef Assessment and Monitoring Program
CSA	Colony surface area
DINAB	Portuguese acronym for National Directorate of Environment
EGM	Ecological Gradient Model
ESR	Environmental Status Rank
ICRI	International Coral Reef Initiative
LC	Live (coral) cover
LSA	Live (coral) surface area
LT	Live (coral) tissue
MERCI-Cor	French acronym for Method to Prevent, Reduce and Offset Impacts on Coral
MPA	Marine Protected Area
MTA	Portuguese acronym for Ministry of Land and Environment
NI	Nature Index
NNL	No net loss; an appraisal of the impact on biodiversity used in environmental mitigation that implies there is a "like for like" net outcome in the offsetting of an activity that impacts the environment. Offsetting actions are required to preserve the same biodiversity in an equal or greater amount to what is lost or impacted.
RAT	rapid assessment transects
RHI	Reef health index
RSM	Reference Site Model
TC	Total (coral) cover
TSA	Total (coral) surface area
WCS	The Wildlife conservation Society
WIO	Western Indian Ocean

Background

The Wildlife Conservation Society (WCS) was established in 1895 and its mission is to save wildlife and wild places worldwide through science, conservation action, education, and inspiring people to value nature. WCS works in more than 60 nations and has supported governments and communities worldwide to create or expand 268 marine and terrestrial protected areas. Mozambique is in one of the 14 regions where WCS focuses on coral reefs and associated organisms together with Eastern Africa, Madagascar, and the Western Indian Ocean (WIO). WCS established a country program in Mozambique in 2012.

The WCS Global Marine Program is investing in ocean protection, sustainable fisheries, and marine species conservation in waters of 23 countries from all five oceans. The Western Indian Ocean is one of the regions for which WCS has developed a marine strategy. The Mozambique marine program started its activities at the end of 2018 and is aligned with that strategy. It prioritises climate resilience and protection of key marine species and key marine habitats. These include corals and fish in coral reef habitats; sharks, rays and their habitats; and marine mammals and their breeding and migration sites. WCS efforts will help to ensure that local communities have continuous access to the natural resources and ecosystems services on which they depend for their livelihoods.

In 2016, WCS Mozambique started the Conservation, Impact Mitigation and Biodiversity Offsets project (COMBO) that envisages to support the Government to reconcile economic development with the conservation of biodiversity, through contributions to improve policies, legislation, knowledge, developing technical tools and providing training on the adequate application of the mitigation hierarchy. As well as a strong focus on developing tools to identify avoidance areas, COMBO works with the National Directorate of the Environment (DINAB), within the Ministry of Land and Environment (MTA), and the Foundation for the Conservation of Biodiversity (BIOFUND) to develop and operationalize biodiversity offsetting protocols for ecological mitigation in line with Environmental Impact Assessment regulations. A key action is to provide technical and scientific advice that identifies metrics to quantify losses of biodiversity that arise from impacts of development projects and gains that are achieved through biodiversity offsetting actions. Coral reef ecosystems are one of the priority biodiversity components identified in Mozambique, due to their high importance for biodiversity, the valuable ecosystem services they provide mankind and their vulnerability to climate change.

A brief introduction to coral reefs

Coral reefs ecosystems cover approximately 250 000 km^2 of shallow tropical to subtropical waters where light attenuation, temperature, salinity, nutrients, and aragonite saturation state enable the growth of corals (Kleypas et al. 1999, Burke et al. 2011). The physical structure of a coral reef is built up over hundreds to thousands of years by the accumulated deposition of calcium carbonate skeletons and consolidation of reef structure by reef-building corals, primarily from the order Scleractinia, together with calcareous material and consolidation from macroalgae (Veron and Stafford-Smith 2000, Fong and Paul 2011). Coral reefs are the largest biological structures on earth (Veron and Stafford-Smith 2000), but are also subject to physical and biological erosion (Scoffin 1992, Hutchings 2000, Schönberg 2008, Carballo et al. 2013, Rice et al. 2019).

Coral reefs provide numerous ecological functions and services and are of high financial value to humanity. Benefits include food and livelihoods associated with local fisheries and tourism, protection of coastlines and human settlements from wave action and storms, sources of medicinal products (Cinner 2014, Spalding et al. 2017). Coral reefs protect the coastlines and support tourism and fisheries of over 100 countries and it has been estimated that as many as 850 million people live close enough to directly benefit from coral reefs (Burke et al. 2011). Although they only occupy approximately 1% of the marine environment, the biodiversity of coral reef ecosystems is high, with suggestions of as many as 950,000 (±40%) multicellular species (Hoeksema 2017), and it is estimated that only 10 % of the diversity has been described (Fisher et al. 2015). This may represent as much as 25 % of marine species (Burke et al. 2011), and includes approximately 4000 species of marine fish (Froese and Pauly 2020) and approximately 800 species of zooxanthellate scleractinian corals, which are mostly reef-building (hermatypic) and live in warm, shallow water (Veron and Stafford-Smith 2000, Veron 2015, Hoeksema and Cairns 2021, www.coralsoftheworld.org). Integrity of coral reef condition and resilience of the ecosystems are critical to ensure they continue to function and deliver ecosystem services sustainably (ICRI 2020a).

Coral reefs are interconnected with other habitats that are equally important in providing ecosystem functions and services, such as mangroves and seagrass meadows. Many of the organisms on coral reefs spend part of their life cycle in seagrass or mangrove habitats (Unsworth et al. 2008, Kimirei et al. 2011, Guannel et al. 2016). Mangroves in particular play an important role in regulating water quality, by regulating turbidity and sediment regimes (Victor et al. 2004, Guannel et al. 2016). Turbidity can be deleterious to corals and reef habitats by reducing light levels that reach a reef with consequences for coral species composition and reef extent. Sediment can smother corals (Rogers 1990, Weber et al. 2012), reduce herbivore consumption of macroalgae (Tebbett et al. 2017a, 2017b) and facilitate transport of nutrients and pollutants with indirect impacts on the development of microbes, phytoplankton and survival of the larvae of predators such as COTS (Birkeland 1982, Garren and Azam 2012, Pratchett et al. 2014).

The impacts of human activity on coral reefs, mangroves and seagrass meadows are diverse. Development and industrial activity in particular in coastal and near reef areas may result in deleterious impacts to coral reefs from sedimentation, land reclamation, dredging, mining, coastal defense installations, waste discharge, pollution, ship groundings, oil and chemical spills, and introduction of alien species (Chabanet et al. 2005, Burke et al. 2011). Human impacts have the potential to act in parallel and synergistically to degrade coral reefs (Hughes and Connell 1999, Bellwood et al. 2004, Muthukrishnan and Fong 2014). However, the individual stresses associated with human activity can potentially be managed and even removed over relatively short timeframes.

Coral reefs in Mozambique

Country overview

The diversity of coral reef habitats in Mozambique results from latitudinal and temperature gradients, coastal geology, river systems and coastal islands, oceanic current eddies in the lee of Madagascar, and depth gradients ultimately associated with deep water in the Mozambique Channel. Coral reefs in northern Mozambique host approximately 300 species of zooxanthellate corals (Obura 2012, Veron 2015), and the area of the Northern Mozambique Channel which extends to northern Madagascar represents a hotspot with the highest coral diversity in the Western Indian Ocean, second to the coral diversity of the Coral Triangle (Ateweberhan and Mcclanahan 2016, Obura et al. 2019b). This hotspot extends southwards at least to Pemba and Nacala, south of which fewer surveys have been undertaken (Obura 2012, McClanahan and Muthiga 2017, Obura et al. 2019b).

Coral reefs are abundant north of Quelimane, with noted coral reef environments in the Primeiras and Segundas Archipelago, Mozambique Island, Nacala, Pemba, and the Quirimbas Archipelago. Most of the northern coral reefs are fringing reefs, and a shallow continental shelf is only a few kilometers wide, then slopes steeply from ~50 m to >1000 m (Obura et al. 2019b). Reefs are exposed to seawater circulation from eddies in the channel, accentuated by eddy interactions with slope bathymetry and canyons (Ménard et al. 2014), and this supports higher oceanic primary production and ecosystem functioning (Obura et al. 2019b). The Zambeze River creates a natural barrier to coral reef development which limits connectivity of coral reef communities, from North to South and contributes to lower coral reef diversity in southern regions of Mozambique (McClanahan and Muthiga 2017).

Coral reefs in and near the Bazaruto Archipelago as well as coastal areas near Xai-Xai, host a zooxanthellate coral diversity of approximately 100 species (Schleyer and Celliers 2005, Veron 2015). In the most southern regions of Mozambique corals become less diverse and more isolated, frequently present as coral communities that do not build reefs and may be intermittent, and which establish on submerged rock, rubble or sandstone substratum from fossilized dunes. This is seen at Inhaca Island and marginal coral communities that establish on isolated reefs as the marine environment becomes progressively subtropical (Schleyer and Pereira 2014).

Threats to coral reefs in Mozambique

The coral reefs of Mozambique have repeatedly been subjected to widespread environmental stresses in recent decades. Bleaching events have impacted the corals of Mozambique and were described to variable extents in 1998, 2004, 2005, 2010, and 2016-7 (Wilkinson 2004; McClanahan, Maina, and Muthiga 2011; Obura et al. 2017; Gudka et al. 2018). Cyclones have historically disturbed coral reefs of Mozambique and are predicted to become increasingly frequent in this region in association with La Niña warm sea surface anomalies (Vitart et al. 2003, Fitchett and Grab 2014). There are also scattered reports of crown-of-thorns starfish (COTS), *Acanthaster sp.*, affecting reefs in Mozambique (Wilkinson 2004, Haszprunar et al. 2017). COTS outbreaks were reported for reefs in the Bazaruto Archipelago and nearby coastal reefs in 1995-1996, impacting approximately 90% of corals on affected reefs (Schleyer and Celliers 2005). More recently COTS were reported from reefs in southern Mozambique (Celliers and Schleyer 2007). Hill et al. (2010) also report COTS in the Quirimbas Archipelago of northern Mozambique and COTS outbreaks have also disturbed reefs in southern Tanzania near the Mozambique border (Wagner 2007).

Human activities have also impacted coral reefs in Mozambique during recent decades. Population migrations in Mozambique driven by armed conflict between 1976 and 1992, extreme weather events generating floods and droughts, and environmental degradation, have all frequently increased coastal populations and pressure on coral reef environments (Raimundo 2009, Menezes et al. 2011, Stal 2011, Blythe et al. 2013). Artisanal fishing has frequently applied destructive fishing gears, exacerbated by migrant fishermen, on reefs and other marine environments along the coast, typically including beach seines, gillnets, and mosquito nets, frequently damaging habitats, exploiting juvenile fish populations and impacting corals (Wilkinson 2004, 2008, Menezes et al. 2011). The human population and the number of fishers and collectors has also increased steadily in Mozambique since 1965, increasing the pressure on marine environments (Jacquet et al. 2010). Industrial developments, including port expansions, mining and oil and gas extraction, create impacts on marine and coastal environments that extend the full length of the coast of Inhambane or are more localised in locations such as Moma, Angoche, Nacala, Pemba, and Palma (Quirimbas Archipelago), with more developments proposed (Pereira et al. 2014).

There is currently a need for guidelines for the ecological mitigation of human impacts to coral reefs, as well as a need for baseline knowledge that can be used to assess human impacts on many of the marine environments of Mozambique (Pereira et al. 2014). It is widely believed that greater control of direct human impacts on coral reefs is critical to enabling coral reefs to withstand the currently less controllable stresses from ocean warming and acidification and climate change (Hughes et al. 2003, Carilli et al. 2009). Here we introduce the concept of environmental mitigation for coral reefs, identify indicators that can be used to describe the condition of a coral reef as well as metrics and methods to measure indicators, and identify indices to combine the insight from multiple indicators in coral reef ecosystems. The objective is to develop a guide to assess the condition of coral reef ecosystems at sites where an impact is planned as a result of a development project, and at sites where these impacts will be offset to achieve no net loss or net gain of biodiversity.

An introduction to ecological mitigation

Ecological mitigation is a hierarchical approach, of seeking to avoid, minimise, restore and, in the last instance offset, to achieve a no net loss or ideally a net gain of biodiversity and ecosystem function when an ecosystem is faced with an impact. Ecological mitigation is frequently associated with large-scale development projects that result in damage or loss of habitats and biodiversity.

Mitigation choices will be influenced by the nature of the impacts to a coral reef ecosystem, which can vary as a result of their source and duration. An impact may result directly from action at a location, or indirectly from action elsewhere such as stresses to interconnected ecosystems. For example, the clearing of mangroves may result in lower sediment trapping from terrestrial run-off and in greater sedimentation on nearby coral reefs. Indirect impacts are likely to be more difficult to quantify, and it may be more appropriate to describe these qualitatively. Impacts are also temporary or permanent in nature. For example, the sediment and turbidity caused by a dredging activity for a port or shipping channel may have temporary and indirect impacts on a nearby coral reef by reducing the light received by corals and their photosynthetic symbionts. Whilst, the impacts are direct if sediment smothers corals. More permanent direct impacts are also likely at the dredged location and the dumping sites of dredge spoil.

The hierarchical steps in an ecological mitigation process may be defined as:

- Avoid any impact(s), in which alternatives are identified to attempt to modify the proposed project and avoid impacts.
- Minimise the impact(s), when after all avoidance measures have been implemented and project-related impacts still exist, then attempts are made to reduce the magnitude, intensity and duration of the impacts.
- Restore biodiversity at sites where impacts have occurred.

- Offset the residual impact(s), when non-negligible impacts [1] still exist after all opportunities to avoid and minimize these impacts have been exhausted, and restore affected biodiversity, this approach seeks to "replace" the biodiversity and ecosystem function elsewhere. The impacts must be evaluated for these to be offset by activities to protect equivalent habitat and biodiversity to that lost or degraded by a project. Ethically, offsetting should be considered as a last resort given the current global and local threats to coral reefs, as well as the limitations of mankind's' ability to manage and re-create nature.

Offsetting of coral reefs may rely on restoration approaches and the establishment of effective marine protected areas (MPAs) to recreate biodiversity value elsewhere in an attempt to replace the biodiversity that is destroyed at the site of impact. However, offsetting of a coral reef does not necessarily require restoration efforts, which are targeted at improving the condition of a degraded coral reef ecosystem. It is assumed that restoration or recovery is timely and predictable when it is applied, however complex ecosystems may require long timeframes and have a low certainty for successful offsetting. An important distinction between offsetting and restoration is that offsetting seeks to replace equivalent biodiversity to that which is impacted by a specific project, whilst restoration seeks to return the biodiversity of a degraded ecosystem to its natural and historic trajectory. We provide a brief overview of coral reef restoration in Box 1.

[1] A non-negligible impact is a direct or indirect impact on biodiversity that must be offset, and that is a result of the impacts of a project within its direct or indirect area of influence, and which is foreseen to exist regardless of the adequate application of efforts to prevent, minimise and restore in line with practices of ecological mitigation.

Box 1. Coral Reef Restoration

Restoration is "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed and attempts to return an ecosystem to its historic trajectory". Restoration aims to eventually leave the ecosystem in a way that requires no further intervention or assistance and a restored ecosystem "contains sufficient biotic and abiotic resources to continue its development without further assistance or subsidy". Restoration is often indistinguishably confused with the rehabilitation of a coral reef ecosystem (Boström-Einarsson et al. 2020). The rehabilitation of an ecosystem is the reparation of ecosystem processes, productivity and services, without the requirement of a return to pre-existing biotic conditions, and often requires attendance.

The defined objectives differ between coral reef restoration projects (Hein et al. 2017). This may be to accelerate reef recovery post disturbance; re-establish a self-sustaining and functioning reef ecosystem; mitigate anticipated coral loss prior to a known disturbance; reduce population declines and ecosystem degradation; provide alternative sustainable livelihood opportunities; or promote coral reef conservation stewardship.

Restoration of coral reefs is currently applied with relative success to spatial scales of the order of 100 m^2 and may be undertaken passively or actively (Boström-Einarsson et al. 2020). Passive restoration, also referred to as 'natural regeneration' or 'indirect restoration', relies on the removal of factors that cause stress or degradation to achieve increases in the abundance and diversity of organisms, without direct planting or seeding. For example, re-vegetation of riverbanks and coastlines to reduce coastal erosion and reduce sedimentation and turbidity at a nearby coral reef is passive or indirect. Active restoration, also referred to as 'direct restoration', relies on directly manipulation to stimulate recovery, such as reintroductions, augmentations or removals of biota. The latter is frequently referred to simply as 'restoration'. Examples of active restoration may include augmenting a coral community with fragments raised elsewhere (Williams et al. 2019, Boström-Einarsson et al. 2020), removal of crown-of-thorns starfish (Babcock et al. 2016, Dumas et al. 2016, Westcott et al. 2020), or removal of macroalgae (Ceccarelli et al. 2018).

Several shortcomings of coral reef restoration projects have been identified (Boström-Einarsson et al. 2020). These shortcomings include:

1. The restoration efforts are limited to short timeframes and small areas. Coral reef restoration projects rarely monitor ecosystem performance for longer than 12 months, and success is considered poor for scales above 100 m^2 (Hein et al. 2017, Boström-Einarsson et al. 2020).
2. The objectives and metrics used to monitor restoration projects are often poorly defined and inconsistent (Hein et al. 2017, Boström-Einarsson et al. 2020).
3. The design of the monitoring undertaken to assess restoration projects is inadequate. Many projects do not include experimental controls, adequate replication, or choose poor controls or references against which to compare progress (Hein et al. 2017, Boström-Einarsson et al. 2020). Control sites should be nearby degraded but unrestored reefs to provide a reference to distinguish between the effects of intervention versus natural recovery (i.e. no treatment effect). Reference sites for what the restoration target is may be rare in areas with low environmental protection, thus best practice is to build empirical models based on information on specific ecosystem attributes obtained from multiple reference sites that are environmentally and ecologically similar to the project site but optimally have experienced little or minimal degradation (Gann et al. 2019). An ideal reference model describes the approximate condition the site would be in had degradation not occurred.
4. The monitoring of coral reef restoration projects is not planned to be ongoing, and rarely undertaken for more than 12 to 18 months (Boström-Einarsson et al. 2020).
5. The progress and outcomes of restoration projects is poorly or not reported (Boström-Einarsson et al. 2020).
6. Coral reef restoration project does not monitor success with a variety of indicators that represent the condition of the ecosystem. Many monitor only coral fragment survival and growth as indicators, and few combine this with other ecological factors (Hein et al. 2017). Hein et al. (2017) suggest a minimum of six ecological indicators to capture the effectiveness of a coral restoration project (coral diversity; herbivore biomass and diversity; benthic cover; recruitment; coral health; structural complexity).

A general introduction to the use of indicators

What is an indicator?

For ecologists the term “indicator” refers to a natural variable that describes ecological condition, responds to environmental modification and may be representative of a delimited area, population or community (Noss 1990, ten Brink 2000, Feld et al. 2009, Certain et al. 2011). Assessing the condition of a coral reef ecosystem or community, as well as assessing impacts or changes in this condition, requires an ability to measure indicators for the environment or community of a coral reef that communicate information about the condition of the ecosystem or community in a simplified and useful manner. Indicators are used to describe a condition related to a context of interest, which responds to specific modifications, and which may be compared to a reference or control state of condition.

Confusion may arise around the term indicator as this term may appear to be interchanged in the coral reef literature, for example with the terms “bioindicator” used to refer to biological indicators as opposed to physical indicators (Chabanet et al. 2005), “metric” which refers to the measurement of an indicator (Nash and Graham 2016), “ecological factor” which may is represented by an indicator (McClanahan et al. 2012b). Authors also often qualify indicators by specifying the nature of interest, for example “resilience indicators” is often used to refer to indicators of the state of resilience of a coral reef ecosystem or community (Obura and Grimsditch 2009, McClanahan et al. 2012b). An indicator is ideally measurable and the term “metric” is used to refer to the units of a particular scale that can be used to measure and express an indicator (Certain et al. 2011, McClanahan et al. 2011a). For example an indicator named “biomass of reef fish” can be expressed in kilograms per area and the metric units chosen could be “kg/ha”.

When they are used effectively, indicators are expected to reveal conditions and trends of an ecosystem or community that support conservation and management decisions (Flower et al. 2017). This often involves comparisons with other similar ecosystems and communities. Expert opinion and empirical evidence has resulted in the identification of indicators that can be used to describe the condition of a coral reef (Jameson et al. 1998, 2001, Jameson and Kelty 2004, Fisher et al. 2008, Obura and Grimsditch 2009, Rodgers et al. 2009, Burke et al. 2011, Flower et al. 2017, Obura et al. 2017, ICRI 2020b). Nonetheless, the data currently collected at regional and global scales are not sufficient to compare all attributes of coral reef ecosystem, community or functional condition (Obura et al. 2019a, ICRI 2020a), which continues to limit the local assessment of coral reef conditions in a wider context.

The purposes of using indicators?

Using indicators to assess the condition or performance of an ecosystem first requires that we define a purpose and the ecosystem. The purpose here is to monitor the condition of a coral reef ecosystem to guide ecological mitigation actions. Indicators vary in their type or focus with examples including ecosystem status indicators, ecosystem threats indicators, ecosystem resilience indicators, or fisheries indicators (Ablan et al. 2004). Indicators have been selected for a variety of purposes associated with coral reefs that include assessing the:

- impacts of climate change on coral reefs (ICRI 2020b),
- resilience, resistance and recovery of coral reef ecosystems (Obura and Grimsditch 2009, Maynard et al. 2010, McClanahan et al. 2012b),
- impacts of fisheries on coral reef ecosystems (Jennings 2005, Nash and Graham 2016, Nash et al. 2016),
- social impacts and food security associated with coral reefs (Hughes et al. 2012),
- human well-being and ecosystem services from coral reefs (Cinner et al. 2009, Ringold et al. 2013),
- effectiveness of management regimes or marine protected areas (Pomeroy et al. 2004, Pelletier et al. 2005, Flower et al. 2017),
- economic values associated with the environment (Pelletier et al. 2005, Spalding et al. 2017).

The selection of the most appropriate indicators for an assessment or monitoring program depends on the objectives of that program (Dale and Beyeler 2001). Chabanet et al. (2005) identify three broad categories of objectives when using indicators to assess condition or change for coral reefs:

- To monitor trends in habitat conditions across time, in order to measure whether specific management actions improved habitats conditions, or whether the habitat has reached a level of disturbance for which some type of actions are required. The monitoring can be specifically designed to address one pre-identified disturbance, or can target a wide spectrum of disturbances. It is wise to predict potential natural (e.g. storm damage, bleaching events) and human disturbances (e.g. illegal fishing, dredging) and include indicators for these in the monitoring to avoid confounding the interpretation of disturbance specific to the planned actions associated with a mitigation project.

- To make a single assessment of the initial condition of the environment. This condition may serve as a reference of habitat conditions after a perturbation has been identified (e.g. ship grounding, storm damage), or draws an initial picture of habitat conditions before some type of planed disturbances occurs (e.g. dredging project, installation of a structure). This objective can be a prelude to the first objective (monitoring).
- To improve knowledge and use of existing indicators or test new indicators. This methodological objective is generally designed to improve the cost-effectiveness of currently applied methods. It aims to test experimentally some hypothesis or it tries to identify hypotheses that will be tested afterwards. As a warning, methods and metrics should not be changed during a project, as this may impede comparison of results at different stages of the life of a project. The improved knowledge is of benefit to future projects.

The need to use a group or set of indicators

Frequently a group or set of indicators is identified to describe as many as possible aspects of the condition of a complex ecosystem or community, such as a coral reef. The choice of indicators can vary from few but widely comparable indicators (ICRI 2020b) to numerous and specific indicators that are collected only in detailed and localized studies (Babcock et al. 2013, 2018). Each new indicator should result in the addition of information of the condition of the ecosystem or community (Certain et al. 2011). Using multiple indicators ensures broad coverage of many aspects of an ecosystem and biodiversity, such as: structure, function, and biodiversity (Noss 1990), and provides opportunity to monitor different environmental pressure or ecosystem services (McClanahan et al. 2012b, Nash and Graham 2016).

It is most effective to use the smallest number of indicators that describe the ecological condition of a coral reef ecosystem or community adequately for the specified purpose(s). It is key to select a core group of indicators that can be used to detect change in a coral reef ecosystem. A greater number of indicators may increase the detail of knowledge of a system, however there is also a risk that too many indicators create a monitoring burden that is not achievable, and leads to monitoring inconsistencies between different parties, as well as to reduced consistency of the data collection and incoherence of knowledge (Reyers et al. 2017). A greater number of indicators may also complicate the calculation of an index, used to interpret of the condition of a coral reef community or ecosystem by increasing the number of unresponsive indicators (McClanahan et al. 2012b)

Grouping indicators to assess condition is ecosystem and context specific (ICRI 2020b), and the context should be defined at an early stage together with the purpose for assessing the condition of an ecosystem or community. For example, Hein et al. (2017) suggest the following six ecological indicators capture the effectiveness of a coral reef restoration project: coral diversity; herbivore biomass and diversity; benthic cover of corals ad other biota; coral recruitment; coral health; and reef structural complexity. Whilst, McClanahan et al. (2012) suggest the following 11 indicators or ecological factors can be used to describe the state of resilience of a coral reef ecosystem: resistant coral species; temperature variability; nutrients (pollution); sedimentation; coral diversity; herbivore biomass, physical human impacts; coral disease; macroalgae; recruitment (of corals) and fishing pressure. Alternatively, Nash and Graham (2016) review indicators in the context of fisheries in a coral reef ecosystem.

Under certain context it may be appropriate to complement the general monitoring of the condition of a coral reef may with specific subsets of indicators with more detailed focus. For example the following set of indicators can be used to assess the condition of a hard coral community in response to a known source of impact (Fisher et al. 2008), such as pollution or sediments from dredging:

- Species (taxa) richness: number of species occurring at a station or location
- Colony density: number of colonies/m^2 sea floor
- Average colony surface area (AvCSA) = ΣCSA/number of colonies
- Colony size coefficient of variation (CSA-CV) = standard deviation CSA/mean CSA
- Total Coral Cover (TC) = TSA/m^2 sea floor
- Average Percent Live Tissue (Av%LT) = Σ%LT/number of colonies
- Live Coral Cover (LC) = Σ LSA/m^2 sea floor
- Percent live surface area3 (%LSA) = [LSA/TSA] × 100

Definitions pertinent to the calculation of the above indicators include: Colony surface area (CSA), which is derived for each colony using a 3D hemispheric surrogate (m^2); Percent live tissue (%LT), which is estimated for each colony; Total surface area (TSA) = ΣCSA, which is the sum of colony surface areas; and Live surface area (LSA) = CSA $\times$ (%LT/100), which is the proportion of a colony surface area that is live. To achieve such a detailed assessment of the hard coral community of a reef, it might be possible to undertake routine (e.g. monthly) observations and photographs of a selection of tagged corals on a reef.

Box 2. Considerations for the choice of individual indicators

The qualities of indicators differ and influence the choice of indicators to detect change resulting from specific stresses against the natural variability of an ecosystem. The first consideration is that indicators chosen should adequately describe, and facilitate the detection of change, in ecosystem or community condition at meaningful levels. As well as appropriate indicators, this requires adequate replication, controls and considerations of the statistical power of the planned data analysis (Hurlbert 1984, Jennings 2005). A second consideration is to identify objective criteria for judging the well-being or ideal condition of an ecosystem, as this will influence the choice of indicators. Thirdly, a choice of what to monitor among the biota and processes occurring in the coral reef ecosystem or community has to be made. For example corals and macroalgae may be considered key organisms and herbivory a key process (McClanahan et al. 2012a). The chosen indicators may have integrative properties that reflect changes in more than one aspect of the system (Garcia and Staples 2000), and for example coral cover is an indicator that reflects the outcome of numerous processes and interactions. However, it is also important to avoid overly simplistic assumptions about the nature of the biological and ecological processes and human and ecosystem behavior.

In line with expert opinion from members of the International Coral Reef Initiative (ICRI 2020b), we recommend choosing indicators that:

- have clear links to the purpose and ecosystem of an assessment,
- have a basis in peer-reviewed literature and quantitative metrics,
- can be assessed at local and regional scales as well by country or management regimes, and are widely used to increase the ability for comparisons between coral reef ecosystems and communities
- are practical and cost effective to implement and provide insight over relatively short timeframes of within months to a few years, so they can be incorporated widely into monitoring frameworks.

Indicators also differ with regards to the insight they provide into the condition of an ecosystem or community. It is useful to consider the information an indicator provides regarding stress, exposure or response, and a variety of indicators with different generic properties should be considered (Ablan et al. 2004), such as:

- Stressor indicators, which measure the stressor itself (e.g., sediments in the water column after a dredging operation). A shortcoming is that these provide no indication of impacts to the coral reef ecosystem or community itself;
- Exposure indicators measure the amount of stressor to which the habitat is exposed (e.g., number of fishing boats operating near a reef, or number of reef-walkers in a tourist area). These could be used as a diagnostic indicator as they are specific to the stressor.
- Response indicators measure condition and can identify changes occurring in the coral reef ecosystem or community (e.g. coral cover measured at yearly intervals); however they do not necessarily identify the cause of the changes. How specific a response indicator is to a stressor is a key criterion.

The sensitivity of indicators and the metrics used to measure them defines their capacity to reveal changes and levels of response to stress and disturbance (Jameson et al. 1998, 2001). A sensitive indicator can provide early-warning signals of change, such as trends in environmental conditions over the time, whilst a less sensitive indicator is retrospective and provides evidence of change in ecosystem or community condition after the change has occurred (Chabanet et al. 2005).

The responsiveness, or timeframe upon which change may be detected, is an important quality that complements sensitivity. A mitigation project is likely to require insight over relatively short timeframes of up to one year, although monitoring over longer timeframes is desirable to evaluate success (Hein et al. 2017, Boström-Einarsson et al. 2020). However, the ability to detect changes sooner allows for management actions to be taken or adapted sooner, and provides quicker feedback on actions taken.

Project design considerations that impact the effectiveness of indicators

For a mitigation study to be successful it is necessary to set reference points or controls, and directions or trajectories for the state of indicators that will satisfy the overall objectives of mitigation. This can be achieved with benchmarks or thresholds, such as for fish biomass, if these have been adequately defined in the scientific literature (McClanahan et al. 2019a). It is also critical to confirm that targets and directions in trends can be met, as well as that limits or thresholds can be avoided.

Past criticism of offsetting projects have included under sampling or inadequate sampling design (Curran et al. 2014, Boström-Einarsson et al. 2020). This highlights the importance of adequately replicating observations of any indicator, because a poorly designed monitoring program or inexperienced observers, may not detect change or statistically defend observed changes, regardless of the sensitivity of the indicators and metric used (Hurlbert 1984, 2004, Barata et al. 2017).

The consideration of scale is also important for a mitigation hierarchy, because this impacts the decisions required to achieve no net loss of biodiversity, habitat and ecosystem function (Chabanet et al. 2005). In the context of coral reefs, the scale of interest may increase from individual coral colonies, through reefscape, reef zone, individual reefs, islands, archipelagos or stretches of coastline up to regions or oceans. Accordingly, scale can be distinguished as:

- Small-scale, when relevant to an individual coral colony, community or reefscape and a spatial unit of 1 m to 10 m;
- Meso-scale, when relevant to a reef zone or whole reef and typically distances (or areas) measured in tens of meters to a few kilometers. Reef zones may be considered large, however they are often influenced by similar environmental or human impacts and respond in a relatively uni-modal way.
- Large-scale, when relevant to reef complexes, islands, archipelagos or regions that can be considered a unit in terms of biogeography or climate. This typically ranges from a few hundreds of meters to hundreds of kilometers.

Quantifying environmental gains and losses

To quantify gains and losses requires a reference to coral reef condition at either a previous point in time or one or more other locations. A widely used approach is the before-after-control-impact (BACI) survey design, which is a method to evaluate natural and human-induced impacts on ecological variables when it is not possible to randomly select the treatment sites (Conner et al. 2016). In this procedure, the control (C) and potentially impacted (I) sites are sampled before (B) and after (A) the development occurs (Underwood 1992). The concept of BACI is that the disturbance in an impacted location will result in changes in the coral reef community from before to after the start of the impact, and these changes can be compared to changes in one or more control locations, which are not impacted and reflect natural change. However, BACI has been criticised because there is a risk that a location-specific temporal difference that occurs between an impact and control locations will be incorrectly interpreted as an impact when this is not a result of the planned human disturbance, therefore BACI is improved when impact and control sites are replicated spatially or temporally (Underwood 1992).

Identifying control or reference coral reef communities is complicated by the immense natural variation observed in coral reef communities and habitats (Rodgers et al. 2009).
One option, which acknowledges control or reference communities are not available from identical communities or habitats at the start of a monitoring project is to assess the "difference in change" observed for coral reef indicators (e.g. fish biomass or coral cover) (Ahmadia et al. 2013). This compares the changes in the ecosystem that occur between managed or impacted coral reefs based on baseline data or the first time point of data collection. If more change is observed between two points in time at the impact site than at other sites, the greater amount of change is attributed to the impact. A benefit of this approach is that it accounts for the inevitable differences between coral reef sites and therefore may be a more accurate assessment of impacts and management (Ahmadia et al. 2013)

Another approach is to use multiple coral reef communities to define potential ranges for each indicator and to assess the condition of a coral reef community against a gradient defined between the maximum and minimum values observed for comparable reef habitats within a region or area (Rodgers et al. 2009). In this approach stratification of habitats is suggested, for example to account for natural variation in communities that is associated with depth or exposure to waves. This somewhat deals with the uncertainty of reference sites representing inherently different communities or habitats. The later approach requires basic mathematical modelling of variables (e.g. linear regression), and becomes more powerful with increasing numbers of sites from which monitoring data is available, and provides the potential to make detailed regional or global comparisons.

Trends versus benchmarks and thresholds

It may be appropriate to consider a positive change in an indicator, which may not necessarily be an increase in the metric (e.g. macroalgae cover), to represent a gain.
It is important that any positive change is record as a trend in positive change over a suitable length of time, thus short term or seasonal fluctuations should not be interpreted over yearly trends. For example macroalgae biomass fluctuates seasonally, with species specific patterns, thus a change in substrate cover measured at two points in time 6 months apart is not likely to reflect the yearly trends in substrate cover(Diaz-Pulido et al. 2009, Lefèvre and Bellwood 2011). Furthermore the trends observed in macroalgae cover are species specific as a result of competitive interactions, and for example trends in *Sargassum spp*. are unlikely to be the same as for *Lobophora spp*. or *Dictyota spp*. which are likely to be shaded by *Sargassum spp.* when these are abundant (Carpenter 1990, McCook et al. 2001, Hughes et al. 2007, Birrell et al. 2008, Lefèvre and Bellwood 2010).

For some indicators, it is possible to define a threshold or benchmark to determine if there is a positive or negative effect on the condition of a coral reef. Numerous empirical studies have suggested thresholds and benchmarks for a variety of indicators of coral reef condition (Jameson and Kelty 2004, Flower et al. 2017). This includes the WIO, where regional benchmarks and thresholds have been suggested for the biomass of coral reef fish in the WIO (McClanahan 2018, McClanahan et al. 2019b), coral cover (Perry et al. 2011, 2013, 2018), and macroalgae cover (McClanahan and Muthiga 2017, Obura et al. 2017). However, it is important to recognise that thresholds may not necessarily reflect the local conditions of a coral reef, and it may be appropriate to build in conservative margins or undertake local research to define local thresholds. In each of the detailed treatments of indicators in this publication we reference the studies from which indicator benchmarks and thresholds were sourced so that it can be determined if local research is desirable, and suggest a margin of caution is used in face of local uncertainty.

Considerations of scale in the assessment of gains and losses

Spatial scale may be considered when quantifying gains and losses (Pioch et al. 2017). It is possible to assess the state of a population of hard corals from a single species, or the state of a community of corals within an area of reef. At an increasing spatial scale the community structure and the habitat complexity of the coral reef habitat can be assessed. So far these population to community relevant scales can be addressed by the indicators suggested in this document which quantify biomass, diversity, functional composition, demographics, relative abundances of competitive organisms and community phase (e.g. coral dominated versus macroalgae dominated), suitable substrate and structural complexity.

It may be equally important is to assess the impacts on nearby seagrass and mangrove communities and habitats, which are likely interdependent with the coral reef habitat, and may be impacted if the coral reef is degraded (Harborne et al. 2006, Guannel et al. 2016). Many organisms rely on each of these interconnected habitats for successive stages of their life cycle. For example, fish may recruit to seagrass meadows, mature in mangroves and relocate to coral reefs as adults (Mumby et al. 2004, Wilson et al. 2010, Evans et al. 2014, Harborne et al. 2015a). Furthermore, predatorial reef fish may rely on feeding grounds in seagrass or mangroves (Harborne et al. 2015b). Therefore, environmental mitigation processes should also assess whether offsets adequately provide equivalent connectivity and consider interdependence of habitats. However, it is beyond the scope of this document to provide detailed guidance for the assessment of the condition of or impacts to seagrass and mangrove communities or habitats, even though many of the same indicators may be applicable. It should also be noted that guidelines to assess the condition of mangroves in Mozambique have been developed (Macamo et al. 2020).

At the largest scale any habitat that is offset should reflect the impacted habitat's regional context of environmental characteristics such as climate, geology, oceanography and human presence (Rodgers et al. 2009). Although it is unlikely that offsetting actions will be undertaken in a location that is distant enough for climate to vary significantly, it is important to consider that physical conditions associate with geology and oceanography may change over short distances and significantly alter the coral reef environment (Chappell 1980, Scoffin 1992). Human settlements are also a significant consideration as their proximity may influence a suit of sub-lethal impacts that range from boat traffic and noise pollution to chemical pollution and illegal fishing (Burke et al. 2011, Cinner et al. 2016, Maire et al. 2016, Smith et al. 2016).

Indexes and their potential to quantify coral reef condition by combining indicators

The indicators assessed provide a quantitative approach to representing individual aspects of a coral reef habitat in a simplified form, however it is necessary to consider multiple indicators simultaneously to grasp the complex nature of the coral reef habitat and the overall condition of the ecosystem. This has led ecologists to propose indexes of coral reef condition (Kaufman et al. 2011, Lasagna et al. 2014). Indexes are calculated from a selection of key indicators suggested to simplify the best available knowledge and represent the overall condition of the coral reef biodiversity and habitat. However, there are few comparisons of indexes of coral reef condition, and it may be difficult to determine the relative quality of indexes (Jameson et al. 1998, 2001, Jameson and Kelty 2004, Díaz-Pérez et al. 2016). Therefore, the choice of which index to apply is likely to often result from the ease of it being adapted to local conditions and reflect the available capacity and resources.

To calculate an index, the included indicators are often scaled and weighted to reflect their perceived importance to the condition of biodiversity or habitat (Certain et al. 2011), however there are also examples of qualitative assessments of indicators to attribute values that can be summed for a score (Hodgson 1999, Jokiel and Rodgers 2005, Flower et al. 2017). Scaling is a means of measuring the difference between observed measures of an indicator and the reference state for that indicator, it also allows comparison of indicators with different metrics or units (Certain et al. 2011, Kaufman et al. 2011, Maynard et al. 2015, Thompson et al. 2020). A standardised index can be a valuable tool to support communication with stakeholders such as developers, environmental managers, policymakers, and the general public (McField et al. 2020).

To calculate an index it is also practical to ensure that all numerical, proportional or percentage scoring of indicators increases in relation to the desirable outcome. In this sense a “0” to “1” scale would reflect 0 as least desirable and 1 as most desirable. This implies that in some instances it is necessary to invert the metric used to describe an indicator. For example, the higher the percentage cover of macroalgae the least desirable this is for the condition of a coral reef, therefore it makes sense to invert values (e.g. 67 % would be converted to 100 % - 67 % = 37 %). Thus the higher the percent cover of macroalgae, the lower the score contributing to the calculation of an index.

We undertook a review of 22 published accounts of indexes and approaches that have been proposed to assess the condition of a coral reef. Some of the approaches focus on elements of coral reef condition such as resilience (McClanahan et al. 2012b, Maynard et al. 2015), coral communities (Fisher et al. 2008, Lasagna et al. 2014, Ferrigno et al. 2016), reef fish communities (Cheal et al. 2008, Loiseau and Gaertner 2015), disturbance history (Aronson et al. 1994) or the potential for reef accretion (Hallock et al. 2003, Pisapia et al. 2017). The indexes range in complexity from simple combinations of 2 to 3 indicators (Kaufman et al. 2011) to mathematical modelling to interpret multiple indicators (Rodgers et al. 2009, Certain et al. 2011, Thompson et al. 2020). Indicators may be quantified with empirically studied metrics (Rodgers et al. 2009) or qualified by expert opinion (McClanahan et al. 2012b, Flower et al. 2017). The indexes of coral reef condition also differ with regards to how they are presented, with some tailored to widespread communication with policy makers (McField et al. 2007, 2020, Kaufman et al. 2011), environmental mitigation of human impacts (Pioch et al. 2017, Jacob et al. 2018), public and stakeholders and others tailored to provide greater detail for environmental management and conservation (Rodgers et al. 2009, Certain et al. 2011). These attempts to describe coral reef condition have been made from local case studies (Rodgers et al. 2009, Houk et al. 2012, Lasagna et al. 2014) or based on global monitoring and observations (Hodgson 1999, Kaufman et al. 2011). We select and compare (Table 1) three examples, which we suggest can include sufficient information on the condition of biodiversity and habitat to be suitable for adaptation to environmental mitigation and offsetting of coral reefs in Mozambique:

- The Ecological gradient model (EGM, Box 3) (Rodgers et al. 2009), a method developed to provides a detailed assessment of coral reef community and ecosystem condition;
- The Nature Index (NI, Box 4) (Certain et al. 2011), a method developed to enable a detailed comparison of the condition of all terrestrial and marine ecosystems;
- The MERCI-Cor Environmental Status Rank (ESR, Box 5) (Pioch et al. 2017), a stepwise framework proposed specifically for mitigation exercises in coral reefs, which as a component suggests an approach for assessing coral reef condition.

An alternative to the use of an index that qualifies coral reef condition with a single numerical value may also be desirable. This is because it is challenging for any single index to adequately represent the complex information contained in the assessment of multiple indicators. Therefore, it may be desirable to complement the use of an index. One option is to present secondary indexes for subgroups of indicators, for example to group the indicators of coral community condition or reef fish condition alongside the overall index. Perhaps, the most useful option to fully communicate the information for each indicator succinctly, without loss of information is to present this visually, with heat maps or graphs, that are colour coded according to the desirable coral reef condition or trends (e.g. green satisfactory or positive and red, undesirable or negative). Examples include the bar graphs, heat map graphs, or the ecological recovery wheel (Figure 1). Tin this way the shape of a graph or colour mapping of an image highlights the condition of a coral reef with reference to as many indicators as measured.

Box 3. The Ecological Gradient Model (EGM) (Rodgers et al 2009).

A descriptive model evaluates the ecological condition of a coral reef based on data collected basic techniques most often used in coral reef surveys. A series of ecological indicators are identified by experts and used to describe the condition of a coral reef. The data is gathered with techniques frequently used in coral reef surveys. The approach allows for comparison of a wide range of coral reefs and is improved by increasing the number of reefs sites from which data is available in a region, therefore it is very useful in regions which have an established monitoring program.

The approach is exemplified for Hawai'i, with data originating from long term monitoring programs (CRAMP monitoring protocol), and augmented with rapid assessment transects (RAT) to increase the number of sites from which data is available. A panel of experts selected the indicators for physical and biological factors. The physical indicators include: habitat class, depth, wave action, wave direction, rugosity or substrate complexity, substrate type (e.g. sand, silt), distance to human population, local precipitation, distance to streams or sources of terrestrial runoff, organic and $CaCO_3$ composition of sediments, grain sizes of sediments. The biological indicators include: total coral cover, cover of 6 key coral species, diversity and species richness of corals, reef fish abundance.

With a view to being objective, the ecological gradient model (EGM) was developed based on a wide range of metrics at numerous sites. The EGM distinguishes 12 coral reef habitat classes, and a range for each metric is established within each habitat class. A model was developed to quantitatively rank reef condition along a continuum. The model was created in Microsoft Excel and calculates where a quantified indicator is positioned along the continuum of values for the metric. This approach permits the operator to alter and define criteria appropriate to a specific question. The operator enters a depth, wave exposure and an assessment value for a single factor or a group of factors into the main menu worksheet. A regional percentile for a particular variable of interest is calculated to evaluate that variable relative to all others in a particular class. For example, the fish biomass at a 5m station in Waikı'kı', O'ahu located in the centre of prolonged, high human activity ranks in the lowest percentile (0%) of all comparable south, sheltered stations (49), between 2.5 and 7.5m.

The study also discusses a "reference site model" (RSM), which uses only reference sites, while the EGM takes advantage of all of the coral reef sites for which data is available. Use of reference sites proved to be subjective and unreliable, especially when multiple factors and multiple sites are involved. However, in some cases the RSM is appropriate in demonstrating severe degradation based on factors such as sediment, coral cover and fish abundance.

Box 4. The Nature Index (NI) (Certain et al. 2011)

The Nature Index provides an integrated ecosystem assessment system that gives information on the state of ecosystems rather than on individual areas. NI allows the comparison of condition of multiple habitats or ecosystems and can be applied to terrestrial, freshwater and marine ecosystems simultaneously. NI also aims to link the environmental assessment process to communication with policymakers, improve data accessibility and operability, as well as use consistent indicator sets and reference points to guide the interpretation of biodiversity and ecosystem status and trends.

The index relies on a network of scientific experts, stressing that biodiversity indicators should be based on expert opinion and may be limited by the availability of information. The resulting set of indicators is supposed to represent the best available knowledge on the state of biodiversity and ecosystems in any given area.

Almost any type of natural metrics can be included within the NI. The case study for Norway includes 178 metrics for marine species, with 18 types of measurement that include: Abundance, area, biomass, breeding success, concentration, catch per unit effort (CPUE), demographic trend, density, EQS, fishing landing statistic, HI/NIVA, NQI1, occurrence, percentage of the reference, population size, relative density, relative rank, Shannon index

Each indicator is scaled relative to a reference value, giving a dimensionless quantity ranging from 0 to 1, where 0 is a completely degraded situation and 1 is an optimal situation for biodiversity. Three methods for scaling indicators are provided. Scaled indicator values can be aggregated or disaggregated over different axes representing spatiotemporal dimensions or thematic groups. Weighting options are also available for indicators. Statistical testing for differences in space or time can be implemented using Monte-Carlo simulations.

The reference state, for each biodiversity indicator, is supposed to reflect an ecologically sustainable state for this indicator. The reference value, i.e., the numerical value of the indicator in the reference state, is a value that minimizes the probability of extinction of this indicator (or of the species or community to which it is related), maximizes at least one measurable aspect of biodiversity of the natural system to which it is related, and does not threaten any measurable aspect of biodiversity in this or any other natural system. The observed and reference states of a given indicator can be estimated from data, either by model prediction or by expert judgment.

The index for each ecosystem or subdivision can be calculated with an R code, which is made available as supplementary material in the form used for the Norweigan case study. The approach allows for the simultaneous consideration of interconnected ecosystems (e.g. seagrass meadows, mangroves) but will require adaptation to coral reefs and Mozambique based upon expert consultation.

Box 5. The Environmental Status Rank (ESR) (Pioch et al. 2017)

The MERCI-Cor approach prescribes a stepwise protocol for assessing the condition of a coral reef.

First the protocol describes general characteristics of the environment using a set questionnaire, then quantifies biophysical condition with set questions to assess losses or gains. The initial measure of indicators is undertaken with field surveys (e.g. transects, quadrats), and suggestions are made for methods, sampling strategies and data analyses. In contrast, the assessment of indicator performance relies on expert opinion. However, an attempt is made to reduce the subjective nature of "opinions" by associating scores with pre-defined descriptions specific to the queries and responses relative to each indicator, to assist the user in attributing a score.

The ranked scoring system is defined with a score ranges (0 - 3):

Rank 0: minimum score 0 to 1 (minimal)
Rank 1: scores of 1 to 4/10 (low)
Rank 2: scores of 4 to 7/10 (average)
Rank 3: scores of 7 to 10/10 (strong)

Each indicator is evaluated:

1. its initial state,
2. its state after impact (post-construction) or compensation (post-measures).

The protocol considers components of site location and landscape (7 questions), physical environment (6 questions), and biological environment (11 questions). The score is averaged for each component. The after scores are subtracted from the before scores to determine if there have been gains or losses. Calculations are also proposed for estimation of the required compensation area.

The following is an example of a pre-formed question and responses for biological environment component.

Question:
Are the coral communities diversified (species richness), characteristic of specific environments (deep, swell, confined, etc.) and do they contain exceptional species (keystone or mutualistic sp., ecosystem engineer, etc.)?

Response options:
0. Few or no coral species are recorded on the habitat. These are mainly pioneer, ubiquitous species with no exceptional characteristics.
1. The habitat has high species richness, but there are few exceptional species such as keystone species
2. The habitat has limited species diversity, but these are characteristic of the specific ecosystems and may contain a relatively large proportion of exceptional species.
3. The habitat has high species richness, contains a high proportion of species characteristic of the specific ecosystems as well as exceptional species."

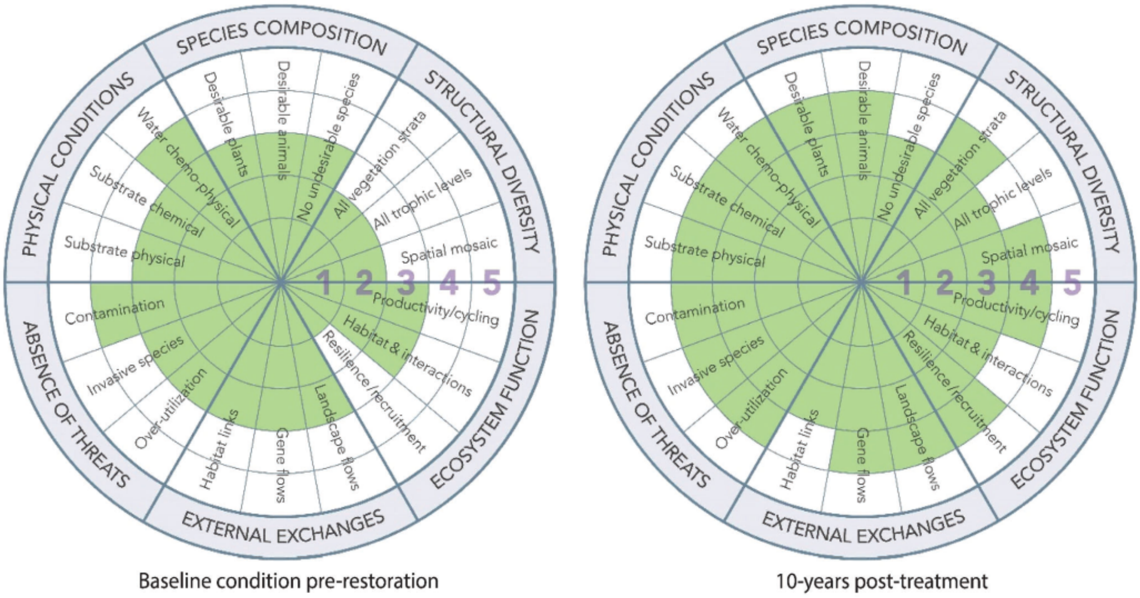

Figure 1. The Ecological Recovery Wheel (McDonald T, Jonson J 2018, Gann et al. 2019). A tool to visually represent ecosystem condition, and communicate the progress of ecosystem attributes or indicators against a baseline or reference model by showing graphs from two points in time (start of project or pre-restoration and after 10 years of restoration actions). Sequential images can show the baseline condition, proposed end condition, and conditions at multiple intermediate points in time. In this example indicators are scored using a score of 1 to 5. The subdivisions and numbers of attributes or indicators may be adjusted to suit the specific project, also using additional colours or patterns.

Table 1. A comparison of the properties of 3 of the methods identified for summarising knowledge of coral reef indicators to assess the condition of a coral reef that are proposed for consideration in this report.

Index	Concept	Number of indicators	Type of calculations	Advantages	Disadvantages	Example outputs
The Ecological Gradient Model (EGM) (Rodgers et al 2009)	A series of ecological indicators are used to describe the condition of a coral reef. A site is assessed against a ranges of values for each indicator from multiple sites in a region.	Approximately 40 indicators are used in the published example when subcategories are included	A proportion (or percentage) is calculated for each indicator to define where the quantified indicator is positioned along the continuum of values for the metric recorded from the region.	The approach allows for comparison of a wide range of coral reefs Uses quantitative metrics of indicators. Data is collected using basic techniques most often used in coral reef surveys. Uses regional ranges of values instead of references to a limited number of control sites There is potential to adjust the number of indicators included to local requirements	Most useful when used in combination with an established monitoring program. The number of sites that have been surveyed for each indicator influences the quality of an assessment. This may be poor if the number of sites is low.	Each indicator is valued as a proportion (or a percentage). 0 (0 %) is equivalent to the lowest record for the indicator in the region 1 (100 %) is equivalent to the highest value recorded for the indicator in the region
The Nature Index (NI) (Certain et al. 2011)	A comprehensive list of indicators is determined and assessed joining efforts of multiple stakeholders to incorporate expert judgment, monitoring-based estimates, and model-based estimates.	The case study for Norway includes 178 metrics for marine species, with 18 types of measurement	Each indicator is scaled relative to a reference value, giving a dimensionless quantity ranging from 0 to 1. Options for scaling or weighting are provided	Accommodates the comparison of different ecosystems (e.g. coral reef, seagrass, mangrove). Experts determine an extensive list of indicators that can be quantified and applied consistently.	Relies on suitable reference sites or models of reference conditions. Applies a large number of indicators and relies on the national or widespread collaboration of stakeholders	A value for each indicator of 0 to 1, where 0 is a completely degraded situation and 1 is an optimal situation for biodiversity.
The Environmental Status Rank (ESR) (Pioch et al. 2017)	Expert opinion is combined with a framework of questions and decisions to standardise application	Components: site location and landscape (7 questions); physical environment (6 questions); biological environment (11 questions).	Answers are ranked on a scale of 0-3 The score is averaged for each component.	Attempts to standardise application and reduce bias in interpretation Relies on expert knowledge and opinion	The set framework of questions and answers, may reduce flexibility to accommodate specific situations.	The after scores are subtracted from the before scores to determine if there have been gains or losses. Calculations are also proposed for estimation of the required compensation area.

Choosing indicators and metrics for a coral reef offset scheme

Method used to identify key indicators and metrics to assess coral reef condition

The context of this document is to propose indicators to assess the ecological condition and function of a coral reef community or ecosystem, so as to guide ecological mitigation decisions and practices. We took the following approach to identify indicators and metrics for coral reefs that can be applied to mitigation projects:

1. Review the scientific basis and global acceptance of indicators and metrics for the assessment of the condition of coral reef ecosystems and communities with a view to applying these to ecological mitigation assessments and offsetting schemes;
2. Identify a group of indicators and metrics that are both complete and effective in their ability to assess the condition of a coral reef ecosystem or community. Prioritising indicators that are either already applied to coral reefs in the Western Indian Ocean Region and Mozambique, or that can be implemented effectively in a relatively short time frame. Whilst, also consider the current data that is available for reference and existing human capacity and resources;
3. Define methods to measure the ecological condition of a coral reef ecosystem or community, including the specific metrics for each indicator, to standardise and guide best practice for the assessments that support ecological mitigation and offsetting decisions for coral reefs;
4. Evaluate the expertise and practical requirements of the methods identified in 3 to assess the current potential for their application in Mozambique, as well as the required capacity development;

A suggested list of indicators of coral reef ecological condition in Mozambique

We identified 101 indicators from our literature search that have been used to evaluate aspects of the ecological condition of a coral reef (Appendix 1). However, the application of all of the potential indicators is likely to be neither practical nor necessary. Therefore, we present a subset of 22 indicators, selected based on their appropriateness for the Mozambican context and to enable international comparisons (Table 2 and Appendix 2).

Subsequent sections of this document provide a detailed review of each of these 22 indicators. We explore the background reasons for choosing each indicator, the methods and metrics used for monitoring the indicator, suggested thresholds and benchmarks based in empirical studies or based upon expert knowledge, and the readiness in terms of capacity, available information and logistics for applying each indicator in Mozambique. The same indicator(s) may be relevant to multiple contexts such as assessing coral reef resilience, coral reef condition, or coral reef susceptibility to specific disturbances. A lack of standardised names for indicators can create confusion in the coral reef literature, therefore to avoid confusion regarding individual indicators we identify alternative names that we are aware of or that are present in the literature.

The following criteria were considered during our selection of the subset of 22 indicators:

- Existing information for the indicator;
- The environmental, ecological or biological insight provided by the indicator;
- The logistics and capacity required for data collection and analysis, as well as the cost and difficulty of including each indicator;
- The completeness and overlap of insight provided by indicators

Table 2. Selected indicators of coral reef ecosystem and community condition and the ecological insight they provide. Note, indicators 1, 2, 3, 4, 5 and 22 are correlated and can be measured simultaneously but their independent consideration provides greater insight.

	Indicator	Ecological insight provided by the indicator
1	Live coral cover	Current condition of the coral community
2	Macroalgae cover (upright and fleshy functional groups)	Indication of the degree to which a coral reef system has shifted from a coral dominated state to a macroalgae dominated state
3	Sponge cover (non-cryptic)	Levels of competition for space with corals
4	Algal turf cover	Indication of the degree to which a system: has shifted from a coral dominated phase, is polluted, or is overfished. Also indicative of levels of competition for benthic space with corals or other organisms.
5	Crustose coralline algae cover	levels of competition for space with corals Reef building/cementing
6	Coral species richness (reef building or zooxanthellate corals)	Photic reefs
7	Coral species diversity (reef building or zooxanthellate corals)	Similar to richness but includes abundance in indices.
8	Coral size frequency distribution	Demographic insight impacts to disturbances and recovery potential of coral populations and the community overall
9	Coral growth rate	Timeframe for recovery of coral cover and to rebuild reef structure; competitive strength against other benthic organisms
10	Stress tolerant coral taxa	Resistance to major disturbance events (e.g. coral bleaching, cyclones, sedimentation)
11	Coral recruit abundance	Natural replenishment and recovery of a coral community
12	Coral recruit survivorship	Natural replenishment and recovery of a coral community; bottlenecks to coral community recovery
13	Prevalence of coral bleaching	Frequency of major disturbances to coral reef communities
14	Partial mortality in corals	Sub-lethal stresses to corals
15	Fish species richness	Ecosystem functional redundancy
16	Reef fish abundance, size and biomass	Reef fish biomass, natural regeneration and recovery potential
17	Functional groupings of reef fish	Ecosystem functional redundancy; Ecosystem resilience
18	Size frequency of reef fish	Reef fish populations and community demographics
19	Corallivory	Disturbances and chronic stress to reef building corals
20	Herbivory	Control of competitors for space with corals, coral reef resilience
21	Coral reef complexity	Insight to the biomass, diversity and ecosystem functions that can be supported.
22	Unconsolidated substrate in the reef environment	Provides insight to the amount of substrate that is not suitable for corals and other benthic organisms.

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