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# Appendix II – Detailed descriptions of key indicators of coral reef condition

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# 1. Coral Cover

## Alternative indicator names

Live coral cover,

## Importance

Critical.

## Definition

The proportion of the substrate that is occupied, or covered, by living reef building corals. Reef building corals are generally considered to be members of the Order Scleractinia, with zooxanthellae symbionts, although some other non-scleractinian corals may be included such as members of the Order Mileporina, Stylasterina, Helioporacea and Alcyonacea that also produce hard skeletons (Veron and Stafford-Smith 2000). The percent cover of reef substratum covered by live reef building corals is considered an indicator of a coral community and the condition of a coral reef.

## Background and Reasoning for choosing the indicator in Mozambique

Living coral cover is the most frequently used coral reef indicator for regional comparisons and policy development because this data is the most often available (ICRI 2020). It is widely used as a basic descriptor of the state of a coral reef and to monitor trends in coral reef condition.

Corals are among the major contributors to building reefs, alongside calcareous algae, and areas with high coral cover such as reef crests and slopes are traditionally associated with progressive accumulation of reef structure (Veron and Stafford-Smith 2000, Perry and Larcombe 2003, Kuffner and Toth 2016). Reef development is a balance of the accumulation of structure, such as coral skeletal growth, and the erosion of structure (Scoffin 1992, Hutchings 2000). Live coral cover thus provides insight to a coral reefs potential to continue to develop or persist under erosive forces and changing environments (Perry et al. 2011, 2013, 2018).

The proportion of a reef covered by live coral provides insight to the overall coral reef community condition by determining the abundance of habitat for obligate coral-dwelling species and corallivorous fishes, or species reliant on coral habitat for recruitment (Pratchett et al. 2006, 2018, Cocker et al 2014). Corals are also key to the development of complex three dimensional structure, and diversity of microhabitats, which is directly proportional to the biomass of reef fish and the overall diversity of a coral reef environment (Graham and Nash 2013). Greater complexity also results in greater surface area of a coral reef and more potential for this to be occupied by living sessile organisms such as hard corals.

The range of live coral cover that is characteristic of a reef is defined to some extent by the environmental parameters of the reef, which are likely to determine the minimum and maximum coral cover possible (Perry and Larcombe 2003, Williams et al. 2013). Coral recruitment, survival and growth are all influenced directly by depth, temperature, light, turbidity and sediment, availability of hard substrate to attach to, as well as the frequency of localized disturbances (e.g. waves, storms) (Kleypas et al. 1999). Regional differences in the taxonomic composition and diversity of coral reef communities may also play a role in determining the characteristic range of coral cover of different reefs with similar environmental parameters (Veron and Stafford-Smith 2000, Roff and Mumby 2012).

Coral cover can be described with relatively high confidence by observers with basic skills or from automated analysis of photographs of the reef substrate (English et al. 1997, Hill and Wilkinson 2004, Kohler and Gill 2006). There are also current developments in technologies focused on mapping reef extent and live coral cover, at least in relatively shallow and clear waters, using satellite and aerial imagery (Andréfouët et al. 2006, Bertels et al. 2008, Li et al. 2020).

Criticisms of using coral cover to describe reef condition include that it is a simplistic measure, which does not provide insight to the diversity, structure or demography of a coral reef community (Hughes et al. 2010, Graham et al. 2011). As a single measure in time of the condition of a reef community, coral cover does not provide insight into the dynamic nature of a reef community. For example, knowledge of coral cover alone does not provide an indication of the prevalence of partial mortality or disease in a coral community, gives no insight into the taxonomic diversity of a coral community, the size frequency distribution of coral colonies sizes, the abundance of recruits and provides no information of the proportion of reproductively mature individuals in a coral community. All of which are key to determining the recovery potential of a coral community.

Coral cover is most useful for comparing the same reefs monitored at different points in time. Coral cover fluctuates as a result of local disturbances, for example storms, crown-of-thorns starfish outbreaks, bleaching events, which can quickly kill a large proportion of a coral community (Bellwood et al. 2004, Hughes et al. 2011, De'ath et al. 2012). However, if the disturbance history of a reef is not known then trends in coral cover alone may provide a poor indication of the overall dynamics of the reef (Hughes 1994, Connell et al. 2004).

Certain coral genera as a result of their characteristic morphologies and growth rates have a larger influence on coral cover. For example corals of the genus *Acropora* with table or branching morphologies can occupy a high percentage of the reef substrate, and can grow quickly to cover a barren substrate (Linares et al. 2011, Madin et al. 2012, Gilmour et al. 2013, Baird et al. 2018). Slower growing corals such as corals of the genus *Porites* with massive morphologies require considerably more time to become the dominant contributors to coral cover on the scale of a reef or reef zone, although this may be observed in sheltered and relatively undisturbed environments (Veron and Stafford-Smith

2000, Hughes et al. 2007, Pratchett et al. 2015). Therefore knowledge of coral cover is more useful when combined with knowledge of the coral taxa present and history of disturbance events.

### This indicator is relevant to

Coral reef community condition,
Coral community condition,
Accretion and erosion of coral reef structure

### Benchmarks and thresholds

WCS scientists, based on peer reviewed research and expert opinion, currently consider 30 % cover of hard coral cover to be a conservation threshold necessary to sustain biodiversity and fisheries on coral reefs (Wildlife Conservation Society 2020).

Based on a study of Caribbean reefs Perry et al. (2013) consider 10 % live coral cover to be the minimum for reefs to accrete at a rate that matches the rate at which they erode.

Live coral cover of 50 % or more is necessary for reef accretion to keep up with sea level rises predicted for the Western Indian Ocean by 2050 under RCP4.5 (IPCC 2013, Perry et al. 2018).

Live coral cover of 70 % or more is necessary for reef accretion to keep up with average sea level rises predicted for the Western Indian Ocean by 2050 under RCP8.5 (IPCC 2013, Perry et al. 2018).

Under RPC2.6, reefs may experience lower increases of seawater depth (~ 10 cm) by 2100 which may enable them to withstand predicted stresses. Greater seawater depth of 50 cm above reefs are predicted for many of the Western Indian Ocean by 2100 and this is likely to increase the destructive action of waves, coastal erosion, and sediment movement further stressing reefs (Perry et al. 2018).

Some authors argue sudden changes in coral cover reflect the impact of stressors and general coral reef condition (Graham et al. 2011). From a meta-analysis of coral cover studies, recovery was greater after disturbances that reduced coral cover to < 10 % of benthic cover (but not < 5 %) and was slowest after disturbances that only reduced coral cover to > 20 %. This suggests the recovery potential and condition of coral reefs that loose more than 10 % coral cover from a disturbance event is poor.

### Readiness

Coral cover is possibly the most widely collected metric, and is an indicator that can readily be compared to reefs in most regions of the world (ICRI 2020). Coral cover is also among the more simple measures and can be described by personnel with basic training.

## Methods and materials

Coral cover can be measured quantitatively by an observer on the reef. Frequently used methods include line intercept transects, point intercept transects, and quadrats (based on points or area) (English et al. 1997, Hill and Wilkinson 2004).

Photoquadrats are increasingly frequently used to describe coral cover. Photos can either be taken by a person in the water or using an automated approach, such as an autonomous underwater vehicle (AUV), or a remotely controlled approach such as a drop camera or a camera mounted to an unmanned underwater vehicle (UUV) (Bryant et al. 2017). Image quality usually limits photoquadrats to areas of the order of 1 $m^2$ or smaller, unless multiple images are joined.

To quantify coral cover from photographs, replicated points or areas of the image are described using software such as Coral Point Count with MsExcel extensions (CPCe) (Kohler and Gill 2006)or PhotoQuad (Trygonis and Sini 2012), points or subdivisions overlayed to an image using printed photographs or image software (e.g Photoshop), or semi-automated identification platforms such as CoralNet (Beijbom et al. 2012).

As the image quality of cameras has developed and image stitching software has become available photo-mosaics and 3D maps or models of coral reef substrate have become another approach for capturing detailed images of larger scale, from which coral cover can be estimated along with other information for a benthic community (Lirman et al. 2007, Edwards et al. 2017).

The advantages of technological developments and the lower cost and greater availability of high quality images of the benthic community include collection of data for a variety of metrics (percentage cover, species composition, disease or bleaching incidence), that can be undertaken using point counts, digitized polygons, area subdivisions or other methods used to analyze underwater imagery (Kohler and Gill 2006, Beijbom et al. 2012). Still photography is generally preferred, however approaches include video transects (Hill & Wilkinson 2004) and mosaic images created from videos (Lirman et al, 2007).

Coral cover is easiest to measure when reef substrates have low three dimensional complexity as the cover of corals is hard to measure in crevices and underneath upright corals. Therefore measures of coral cover are frequently simplified or two-dimensional descriptions of the substrate. This may change as photo mosaic imagery and photogrammetry increase the ability to quantify three dimensional structure accurately (Edwards et al. 2017). Interpretation of coral cover is benefitted when combined with even basic descriptions of coral reef complexity, which can be assessed with qualitative visual assessments or quantitative methods (Hill and Wilkinson 2004, Wilson et al. 2007).

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## 2. Macroalgae Cover

### Indicator Name

Macroalgae cover

### Alternative indicator names

Algae cover, Fleshy algae cover, Upright algae cover, Seaweed cover, Macroscopic algae cover

### Importance

Critical.

### Definition

The proportion of the substrate that is occupied, or covered, by living macroalgae. The percent cover of macroalgae is a general measure of the relative abundance of macroalgae in a reef community.

Macroalgae are multicellular marine plants that are easily observed by the unaided eye and which have a plant body or thallus that is characterized by holdfasts for attachment, and by laminae, reproductive sori, gas bladders, and/or stipes. Macroalgae lack the various structures that characterize higher plants, such as true leaves, roots, and encased reproductive organs (Littler and Littler 2011). However macroalgae can be distinguished into functional groups based on anatomical characteristics (Steneck and Dethier 1994).

### Background and Reasoning for choosing the indicator in Mozambique

Macroalgae are widely present on coral reefs and compete with corals and other organisms for space (McCook et al. 2001, Littler and Littler 2007, Fong and Paul 2011). The ecological interaction of macroalgae are largely determined by their physical features which range from from larger upright macroalgae to diminutive filamentous algae assemblages (Steneck and Dethier 1994, Goatley et al. 2016). Grouping macroalgae according to morphologies and functional attributes simplifies the study of ecological interactions (Steneck and Dethier 1994), although it may overlook the chemical properties of interactions, and is particularly relevant to competitive interactions with corals (McCook et al. 2001, Steinberg and de Nys 2002, Birrell et al. 2008, Diaz-Pulido et al. 2010, Bonaldo and Hay 2014).

Greater macroalgae cover is frequently associated with the degradation of coral reef communities (Hughes 1994, Bellwood et al. 2004, Littler et al. 2006, Hughes et al. 2007, Obura et al. 2017). Macroalgal cover can increase dramatically following disturbances that cause widespread coral mortality and create surfaces for macroalgae to colonise (McClanahan et al. 2001, Littler and Littler 2007, Diaz-Pulido et al. 2009). Increased macroalgae cover is often associated with reduced levels of herbivory which may result from overfishing or diseases that remove reef fish or invertebrates (e.g. echinoderms) (Hughes et al. 2007, Littler and Littler 2007). The pollution of reef waters with increasing levels of nutrients also promotes macroalgal growth and contributes to greater macroalgae cover (e.g. sewage runoff) (Lapointe 1997, McCook 1999, Lapointe et al. 2004, Littler et al. 2006).

Macroalgae presence on a coral reef provides insight to the resilience of a coral reef ((McManus and Polsenberg 2004, Adam et al. 2015, Holbrook et al. 2016)McClanahan et al 2012). Macroalgae are among the primary competitors with coral for space (McCook et al. 2001). There are numerous reports of degraded coral reef ecosystems experiencing a 'phase shift" to macroalgae instead of coral dominance (Hughes 1994, Hughes et al. 2007, Ceccarelli et al. 2020). However, the time that is required for macroalgae phase shifts to develop on coral reefs is regionally variable (Roff and Mumby 2012). Cases of observed coral decline and macroalgae phase shifts have been more frequent in the Caribbean than in other regions (Connell 1997, Graham et al. 2011, Roff and Mumby 2012).

Macroalgae can have both positive and negative impacts on coral reef communities (Birrell et al. 2008, Ceccarelli et al. 2018). The taxonomic composition of a macroalgae community as well as the physical structure, chemical properties and biomass of an algae community influence the success of coral recruitment and survival of established corals (McCook et al. 2001, Birrell et al. 2008, Diaz-Pulido et al. 2010, Bonaldo and Hay 2014), the rates and extent of fish herbivory (Green et al. 2009), reef fish recruitment (Evans et al. 2014), reef fish predation (Madin et al. 2011, Catano et al. 2016, Fong et al. 2018).

Knowledge of the cover of Macroalgae is a useful indicator of the state of a coral reef community. However, the usefulness of knowing macroalgae cover is greatly improved when accompanied by other indicators such as the taxonomic richness of the macroalgae community, height and functional forms of the macroalgae present (Steneck and Dethier 1994, McCook et al. 2001, Birrell et al. 2008, Ceccarelli et al. 2018). Furthermore, macroalgae biomass is predominatly seasonal and the use of macroalgae surveys to compare reef condition is hindered if surveys are undertaken at different times of the year (Lefèvre and Bellwood 2010, Wilson et al. 2014).

## This indicator is relevant to

Coral reef community condition,
Coral reef history of disturbance events,
Coral reef herbivore communities,
Coral reef community resilience,
Overfishing for coral reefs
Pollution and nutrient enrichment of coral reef ecosystems

## Benchmarks and thresholds

WCS surveys of macroalgae cover on reefs of the SWIO suggest that the average macroalgae cover on coral reefs in good condition is 5 % (McClanahan and Muthiga 2017). WCS surveys in Mozambique have suggested the average macroalgae cover on coral reefs is closer to 15 %, which reflects the fishing of herbivores (McClanahan and Muthiga 2017, Birrell et al. 2020).

Obura et al (2017) report a regional average for fleshy algae cover of 14.2±2.5 % for between 1994-1998, and that his increased temporarily to as much as 50 % following widespread coral bleaching in 1998, subsequently stabilizing at 36.3±0.7 % between 1999-2015.

Coral recruitment decreased by more than 80% (from an average of 31 to 5 recruits $m^{-2}$ per year) in Jamaica, when macro-algal abundance increased from 2 % to 20 % cover (Hughes 1985 cited in Hughes et al. 2010).

Macroalgae dynamics in the Caribbean differ from elsewhere, and macroalgae cover on heavily fished reefs in the Pacific (Hawaii and The Philippines) is closer to 20 %, whilst reefs with intact fish communities have macroalgae cover of < 5% (Roff and Mumby 2012 and references therein).

Therefore we suggest that for tropical coral reefs in northern Mozambique:

- 5 % macroalgae cover represents a reef in good condition.
- 5 to 15 % macroagae cover represents a coral reef that shows a certain degree of degradation or overfishing, particularly of herbivores
- 15 % macroalgae cover represents a reef that has been degraded, either by fishing which has removed herbivores, pollution which has increased the seawater nutrient concentrations, or other stresses which have resulted in coral mortality (e.g. crown-of-thorns starfish, coral bleaching)

## Readiness

Macroalgae cover is a widely collected metric, and is an indicator that can readily be used to compare reefs in most regions of the world. Macroalgae cover is among the more simple measures of a reef community and data can be collected by personnel with basic knowledge and training.

## Methods and materials

Macralgae percent cover can be measured quantitatively in similar ways to other benthic organisms and usually is measured using the methods described for organisms such as corals. Methods may include line intercept transects, point intercept transects, and quadrats (based on points or area) (English et al. 1997, Hill and Wilkinson 2004) as well as photoquadrats, videos or photomosaics (Edwards et al. 2017).

Surveys of macroalgae should consider that macroalgae biomass and cover is likely to vary seasonally and that this is specific to taxa (Lefèvre and Bellwood

2010). Storms are also likely to dislodge and fragment macroalgae (Cheroske et al. 2000) and surveys undertaken soon after storms may underestimate the cover and biomass of macroalgae.

More detailed surveys of macroalgae should consider gathering data that include macroalgae height, functional group, taxonomic detail, spatial density of macroalgae stems (Steneck and Dethier 1994, McCook et al. 2001, Birrell et al. 2008, Connell et al. 2014, Ceccarelli et al. 2018). For such purposes surveys may be facilitated by using quadrats, belt transects or photographic methods (e.g. photoquadrats, photo-mosaics).

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## 3. Sponge Cover

### Indicator Name

Sponge cover

### Alternative indicator names

Cover of Porifera

### Importance

High

### Definition

The proportion of the substrate that is occupied, or covered, by living sponges. The cover of sponges is a proxy measure of the relative abundance of sponges in a reef community. However, many sponges are cryptic in nature, boring into the coral reef matrix or coral skeletons (e.g. *Cliona spp.*), which can limit their accurate detection without cores or destructive sampling approaches.

### Background and Reasoning for choosing the indicator in Mozambique

Sponges are competitive benthic invertebrates on coral reefs and the abundance of sponges, measured as sponge cover, may be a useful indicator of the state of a coral reef community (Ladd 2019). Sponges frequently compete for space with corals on coral reefs (Connell et al. 2004, González-Rivero et al. 2011), and hinder coral recruitment (McClanahan 2000, Arnold and Steneck 2011, Brandt et al. 2019). The competitive potential of sponges can be physical or associated with chemical metabolites that are lethal or harmful to established corals (Chaves-Fonnegra et al. 2008, Helber et al. 2018, Pawlik and McMurray 2020) and reduce coral recruitment (Brandt et al. 2019).

Sponges are significant contributors to the bio-erosion of coral reefs (Hutchings 2000, Schönberg 2008). However, sponges can also be important for the consolidation loose reef matrix and rubble, providing a temporary stabilization of the reef substrate that allows for the more permanent consolidation of the reef matrix by crustose coralline algae and corals (Wulff 1984).

Sponges are reported to become more abundant on reefs following large scale disturbances that reduce coral cover, such as bleaching, when herbivory also inhibits the potential increase in cover of macroalgae (Aronson et al. 2002). It is suggested boring sponges are becoming more abundant and that sponge bio-erosion is increasingly affecting coral reefs as a result of climate change (Schönberg 2008, Carballo et al. 2013, Fang et al. 2013). However, sponges are also likely to increase in abundance as a result of the decline in abundance of corals and other reef organisms (Pawlik and McMurray 2020).

Degraded water quality is also thought to promote increased sponge abundance (Pawlik and McMurray 2020). The bio-erosion of coral skeletons by sponges on Kenyan reef has been positively correlated to chlorophyll *a* concentrations (chl *a*), a measure of levels of nutrients in coral reef waters (McClanahan and Carreiro-Silva 2012). Whilst, the abundance of bio-eroding sponges was also greater on reefs with higher abundance of herbivorous fish and lower abundance of herbivorous sea urchins (McClanahan and Carreiro-Silva 2012).

Overall, the ease of using sponge cover as an indicator is impacted by the cryptic nature of many sponges (Carballo et al. 2013). The habitat and community zonation patterns of sponges also influence when sponge cover is useful (Pawlik and McMurray 2020). For example, cryptic sponges are more abundant in shallow waters whilst abundance of exposed sponges (e.g. barrel and vase sponges) is greater in deeper waters.

### This indicator is relevant to

Coral reef community condition,
Coral reef community resilience,
Pollution of coral reef waters

### Benchmarks and thresholds

Identifying thresholds for sponge abundance is challenging for several reasons (Pawlik and McMurray 2020). These include:

- The ecology of sponges on coral reefs is relatively poorly studied.
- Transition from coral- to sponge-dominated reefs has occurred primarily in the Caribbean but evidence is localized and limited whilst also occurs more slowly than for macroalgae.
- Degraded coral reefs, in particular from anthropogenic eutrophication, often show higher sponge abundance, however coral cover has usually also decreased as a result of stressors and sponge abundance may have increased as a result of greater substratum availability from coral mortality.

We suggest monitoring trends of change in sponge abundance over time, monitoring coral cover as well, and that increases of > 5 % substratum cover should give rise to concern. Increases in the abundance of boring sponges should also be monitored from fragments or cores of the coral reef matrix or colonies (e.g. branches, or detached colonies).

### Readiness

Sponge cover is a less widely collected metric than coral cover and macroalgae cover, although non-cryptic sponges can be measured simultaneously to other benthic organisms, for example in transect or quadrat surveys. This requires personnel with basic knowledge and field work training, provided sponge identification is limited to key taxa or functional groups. Surveys of cryptic sponges require training in the collection of coral skeleton and reef matrix samples.

Accurate identification of sponges is likely to require post field work observations using a microscope or dissection scope, also requiring personnel with advanced knowledge.

## Methods and materials

Sponge cover, for non-cryptic sponges, can be described similarly to other benthic organisms with the methods described for corals and macroalgae. This will provide a relative measure of sponge abundance that can be compared though time or between reefs of similar environments. Because sponges are often cryptic in nature, at least in shallow coral reef habitats (Pawlik and McMurray 2020), visual or photographic surveys are likely to underestimate the overall biomass of sponges, with bias towards the exposed sponges in a community.

For a more precise survey including boring sponge abundance or biomass destructive sampling is required, for example using cores, colony branch samples, or cross sections of coral skeletons or pieces of reef substratum (Carballo et al. 2013). When destructive sampling is undertaken this should seek to minimize repetitive damage to a site especially in sensitive habitats or for rare corals.

The accurate taxonomic identification of sponges frequently requires inspection of diminutive structures under a dissection microscope (e.g. calcified spicules). Field surveys may be performed by non-specialist by limiting observations to general sponge categories. It is possible to differentiate sponges by morphology, and the morphologies of sponges in coral reef communities of Mozambique have previously been correlated to habitat features such as depth, complexity and seawater currents (Barnes and Bell 2002)

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# 4. Algal Turf Cover

## Indicator Name

Algal turf cover

## Alternative indicator names

Turf algae cover, Filamentous turf algae (FTA) cover, epilithic algae community (EAC) cover, epilithic algal turf

## Importance

Critical

## Definition

The proportion of the substrate that is occupied, or covered, by an algal turf assemblage.

An algal turf is an assemblages of diminutive macroalgae and microalgae that can vary in their species diversity, density, height, morphologies present, nature and quantity of associated sediments and organic matter (Connell et al. 2014). The height of an algal turf assemblage is generally of the order of millimeters to a few centimeters (Steneck and Dethier 1994, Connell et al. 2014).

The density of algal turfs can range from dense aggregations of erect thalli, an intertwined matrix of erect and prostrate branches to sparse filaments on a mostly visible reef matrix (Connell et al. 2014).

Algal turfs frequently contain small red algae, chain-forming diatoms, cyanobacteria, microalgae and juvenile stages of macroalgae (Connell et al. 2014). Thus, an algal turf may include representatives from the algal divisions Cyanophyta, Chlorophyta, Chrysophyta, Phaeophyta, and Rhodophyta.

The morphologies present in an algal turf may be predominantly filamentous but include crusts, and unicells as well as diminutive specimens of other morphologies (Steneck and Dethier 1994). Thus, algal turfs may include diminutive juvenile, cropped or fragmented macroalgae that otherwise develop various more upright morphologies (Connell et al 2014). However algal turfs are distinguished from assemblages of fleshy algae even if they may later develop into one (Steneck and Dethier 1994, Littler and Littler 2007, Connell et al. 2014).

Algal turfs can be inconspicuous and may be incorrectly described as bare space (Smith et al. 2016). This may occur when the density and height of turf algae is low or in studies focused on organisms that are larger and more visually present such as corals, sponges, soft corals or fleshy macroalgae.

Given the variation amongst physical features of algal turfs that are observed on coral reefs it is important to characterize the features of an algal turf (e.g. height, taxa, density) as an approach to defining the algal turf when this is of specific importance to the nature of a study or survey (Connell et al 2014).

## Background and Reasoning for choosing the indicator in Mozambique

Increasingly algal turfs have become associated with the degradation coral reef habitats (Russ 2003, O'Brien and Scheibling 2018). The abundance of algal turfs provide insight to the condition or state of degradation of a coral reef and the resilience of a coral reef ecosystem (Littler and Littler 2007, McClanahan et al. 2012, Roff and Mumby 2012). More importantly it is only recently acknowledged that there is considerable variation in the features of algal turfs, which must be considered to fully understand their impacts on the condition and resilience of a coral reef ecosystem (Connell et al. 2014).

Algal turfs contribute more than other organisms to primary productivity on coral reefs (Adey and Goertemiller 1987). At the base of the food chain algal turfs are an important food source for herbivorous fish and invertebrates (Kramer et al. 2013, Marshell and Mumby 2015), and indirectly influence the spatial distribution and abundance of coral reef fish (Russ 2003). Thus algal turfs play a major role in defining the food web dynamics of a coral reef ecosystem, the rates of biomass transfer and the biomass that can be sustainably harvested (Tootell and Steele 2016, Morais et al. 2020). However the quality of algal turfs as a food source for herbivorous fish differs, for example as a result of associated sediments (Bonaldo and Bellwood 2011, Goatley et al. 2016).

Algal turfs cover large proportions of the substrate on coral reefs creating unique microhabits (Wismer et al. 2009, Smith et al. 2016, Arias-González et al. 2017). Concerningly, the cover of algal turfs has widely increased on coral reefs following widespread disturbances and degradation of coral reefs, for example from coral bleaching events (Aronson et al. 2002, Hughes et al. 2011, Morais et al. 2020). Filamentous algae, a major component of algal turfs, are quick to colonize dead coral surfaces promoting dominance by algal turfs on disturbed and degraded reefs (Diaz-Pulido and McCook 2002). Furthermore, the dominance of a coral reef assemblage by algal turfs may also represent a precursor or intermediate phase to a complete algal phase shift (Bellwood et al. 2004, Littler and Littler 2007, Roff and Mumby 2012).

Algal turfs compete for space on a coral reef with other organisms such as corals, crustose coralline algae and sponges (McCook 2001, Wild et al. 2014, Swierts and Vermeij 2016). Algal turfs hinder coral recruitment, impacting the recovery potential and resilience of a coral community (Birrell et al. 2005, Arnold et al. 2010a, Mumby et al. 2013, Speare et al. 2019). The competitive threat from algal turfs to coral and other organisms can increase as a result of the allelopathic attributes of individual species in the assemblage and the retention of sediments in an algal turf (McCook et al. 2001, Elizabeth and Roy 2004, Jorissen et al. 2016).

Algal turfs naturally trap sediments, by reducing water flow and physically lodging particles, and this increases the ability of algal turfs to form layers that smother other benthic organisms such as coral recruits and crustose coralline algae (Purcell 2000, Romano et al. 2003, Elizabeth and Roy 2004). The nature of the sediments associated with algal turfs differs as a result of their origin, with terrestrial inputs associated with fine, siliceous material, and often with organic material in 'flocs' which are readily transported in river plumes or via re-suspension (Bainbridge et al. 2012, 2018, Bartley et al. 2014, Tebbett et al. 2018). Sediments may also originate from the physical and biological erosion of reef matrixes and organisms giving rise to the coarser, biogenic carbonate sediments frequently characteristic of coral reefs (Gordon et al. 2016, Tebbett et al. 2017a). The sediments characteristic of a coral reef are often correlated to the distance from landmasses and river mouths as well as the local seawater depth which influences re-suspension by wave action (Acevedo et al. 1989, McCook 2001, Bartley et al. 2014, Tebbett et al. 2018)

The nature and quantity of sediments trapped in an algal turf influence the grazing rates and distribution of herbivorous fish, generally deterring grazing when sediments become more abundant (Wilson et al. 2003, Tebbett et al. 2017b, 2017c). Sediments trapped by algal turfs are even suggested to have driven major shifts in the composition of benthic coral reef communities (Arias-González et al. 2017).

## This indicator is relevant to

Coral reef community condition,
Coral reef community resilience,
Coral reef fisheries,
Coral recruitment,
Primary productivity

## Benchmarks and thresholds

Algal turfs > 5 mm in height are increasingly likely to have negative impacts on coral recruits (Birrell et al. 2005, Arnold et al. 2010b, Ford et al. 2018), trap sediment (Purcell 2000, Elizabeth and Roy 2004) and reflect lower levels of grazing (Ford et al. 2018, Santano et al. 2021).

We suggested algal turfs > 5 mm in height are also increasingly likely to have negative impacts on mature coral colonies (McCook et al. 2001, Elizabeth and Roy 2004, Birrell et al. 2008)

Smith et al (2016) suggest algal turf cover of > 50 % represent coral reef ecosystems that are degraded, for which distance to human settlement was used as a proxy.

Further definition of benchmarks for the cover of algal turfs on coral reefs is complicated by the poor definition of what an algal turf is in many studies (Connell et al. 2014).

### Readiness

The percent cover of algal turfs is an easily measured metric that can be collected at the same time as the cover of other benthic organisms is described. However, algal turfs have often been considered less important than corals and larger macroalgae until recently. Algal turf assemblages are rarely defined on the basis of their features (e.g. height, density, taxa present), which limits comparisons between studies (Connell et al. 2014). More concerningly algal turfs with low density and height may be overlooked and classified as reef matrix.

### Methods and materials

Algal turf cover can be quantified using similar methods as used to describe the cover of corals and other benthic organisms, such as the point intercept transect, line intercept transect, belt transects and quadrat methods used to survey corals.

However, the diminutive form of algal turfs and the features that define an algal turf assemblage such as height, density, sediments, taxa present, and morphologies present, or sediments may require more detailed measurements within smaller quadrats (e.g. 10 cm × 10 cm to 25 cm × 25 cm),

It is advisable to use a ruler or calipers to measure algal turf heights, and to make multiple estimates in a sample area.

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## 5. Crustose coralline algae (CCA) cover

### Indicator Name

Cover of live crustose coralline algae (CCA)

### Alternative indicator names

CCA cover, non-articulated crustose algae, crustose algae, nongeniculate coralline algae, nongeniculate corallines

### Importance

Critical

### Definition

Crustose coralline algae (CCA) are red algae (Phylum: Rhodophyta), of the family Corallinaceae that are characterized by a hard thallus resulting from calcareous deposits in cell walls, and which are nongeniculate; or do not have flexible non-calcified joints or genicula (Steneck 1986, Littler 2013). CCA predominantly adhere to the reef substrate to form a crust, which may be millimeters to tens of centimeters thick (Steneck 1986), and may have vertical relief in the form of knoblike features, shelf like structures or minor branching structures (Adey et al. 1982, Littler and Littler 2013)

### Background and Reasoning for choosing the indicator in Mozambique

Crustose coralline algae (CCA) are among the most abundant benthic organisms and frequently dominate the visible and cryptic surfaces of shallow coral reefs (< 30 m) in the photic zone (Adey et al. 1982, Steneck 1986, Adey 1998, Littler and Littler 2013). Crustose coralline algae taxa are often abundant in cryptic environments such as overhangs and crevices, or high wave energy environments such as reef crests (Adey et al. 1982, Steneck 1986, Adey 1998, Littler and Littler 2013).

Crustose coralline algae play an important role in the production, consolidation, and accretion of the coral reef matrix, (Stearn et al. 1977, Chisholm 2003, Weiss and Martindale 2017). CCA grow prostrate on hard substrata and as epibionts on macroalgae, seagrasses, molluscs and other marine animals (Steneck 1986). Crustose coralline algae also contribute to reef sediments when broken down (Scoffin 1992, Hutchings 2000, Tebbett et al. 2017).

Many species of CCA periodically slough or shed their epithelial layer which is an effective antifouling mechanism that reduces the ability of other benthic organisms, including corals and other macroalgae, to colonize CCA surfaces (Steneck 1986, Littler and Littler 1999, Harrington et al. 2004, Vermeij et al. 2011). This effectively contributes to maintaining a mosaic of relatively clear habitat on a coral reef, reducing establishment of upright macrolagae and cover for predators of reef fish, in particular herbivores, and in turn promoting grazing (Madin et al. 2011). Thus, together with live coral, CCA are important in determining coral reef microhabitats and influence the distribution of reef fish in general (Coker et al. 2014, Russ et al. 2021).

Crustose coralline algae are widely considered to be important for the settlement of coral recruits, and whilst some species of CCA, such as *Hydrolython spp*. and *Titanoderma spp*. attract coral larvae and facilitate higher coral settlement than other coral reef surfaces (Raimondi and Morse 2000, Harrington et al. 2004, Ritson-Williams et al. 2010, Arnold and Steneck 2011), this is likely to be largely because they represent early successional substrates, much like biofilms or polychaete tubes surfaces, that present coral recruits with a window of opportunity to establish (Arnold and Steneck 2011). The antifouling mechanisms of many CCA, such as sloughing epithelial tissue layers (Harrington et al. 2004), or associated bacterial communities (Sneed et al. 2015), inhibit the settlement and survival of coral recruits. Crustose coralline algae will compete for space and overgrow coral recruits, although coral survival increases with size, and is higher in exposed habitats (Jorissen et al. 2020). The competitive threat of CCA towards coral recruits is also a function of specific physical traits of the CCA involved such as thallus thickness and growth morphology (Steneck 1986). Therefore it is somewhat important to distinguish the taxa of crustose coralline algae present on a coral reef before assuming all CCA cover represents suitable substrate for coral recruitment, even though CCA may promote other habitat characteristics of a coral reef in good condition.

Overall a relatively high cover of CCA is associated with better coral reef condition (Smith et al. 2016). CCA cover is generally reported to decline, often overgrown by algal turfs, when reefs are polluted, experience high sedimentation (Purcell 2000, McCook 2001, Karcher et al. 2020) or experience seawater acidification (Short et al. 2014, Fabricius et al. 2015). CCA also benefit competitively when the growth of fouling algal turfs and macroalgae is physically removed on a regularly basis by wave action or herbivorous fish (Kendrick 1991, Littler et al. 1995, Cheroske et al. 2000, Purcell 2000).

## This indicator is relevant to

Coral reef community condition,
Coral reef community resilience,
Coral recruitment
Coral reef fish community composition

## Benchmarks and thresholds

Currently not aware of defined thresholds for coralline algae cover.

The benthic cover of CCA is likely to be habitat specific, for example greater in shallow wave swept habitats such as reef crests, or cryptic habitats such as caves and overhangs. Whilst, lower CCA cover is likely in sheltered lagoons or reef slopes. This complicates providing a general threshold for CCA cover.

We suggest:
> 30-40 % CCA cover may represent good condition on exposed coral reef crests.
> 20 % CCA cover may represent good condition on sheltered coral reef habitats such as back reefs and lagoon patch reefs.

Smith et al (2016) suggest 50 % cover of reef building organisms (CCA + hard corals) is indicative of coral reefs in good condition, whilst ~ 25 % represents coral reefs that are degraded by human activities.

## Readiness

The percent cover of CCA is an easily measured metric that can be collected at the same time as the cover of other benthic organisms is described.

## Methods and materials

CCA cover can be measured quantitatively in similar ways to other benthic organisms and usually is measured using the methods described for coral or macroalgae cover. Methods, include line intercept transects, point intercept transects, and quadrats (based on points or area) (English et al. 1997, Hill and Wilkinson 2004) as well as photoquadrats, videos or photomosaics (Edwards et al. 2017).

It is important to recognize that a survey of the exposed CCA on a coral reef, although considered an indicator of coral reef condition, is not a complete survey of the CCA community of the reef because of the cryptic nature of many CCA. Sampling the cryptic CCA will require destructive sampling (e.g. cores or collection of reef matrix samples)

Similar to turf algae, more detailed surveys require closer observation in smaller quadrats (e.g. 10 cm * 10 cm).

The taxonomic identification of CCA generally requires confirmation of anatomical structures in samples using a dissection microscope (Adey et al. 1982, Steneck 1986, Littler and Littler 2013) and therefore only a few species can be reliably identified in field surveys. However, it is of interest to differentiate genera of CCA so as to provide greater insight into the predominant antifouling mechanisms and impacts on coral recruitment.

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## 6. Coral species richness

### Indicator Name

Coral species (taxa) richness

### Alternative indicator names

Taxonomic richness, number of species

### Importance

High

### Definition

Broadly speaking, species richness is the number of species in a community or region, however it may not be possible to enumerate all of the species in a natural community or region (Krebs 2014).

We define coral species richness as the total number of coral species in a coral reef community. For practical reasons that include differential abundance of species, differences in the detection rate of species, and different abilities of observers to detect species, the community should be specified by an area with a defined search time and sampling effort.

Species richness does not require the identification of the species counted, however the insight that can be gained into the condition of a coral reef community is increased with knowledge of the species present.

### Background and Reasoning for choosing the indicator in Mozambique

Coral species richness is a simple measure of the pattern of diversity of a coral community (Devantier and Turak 2017). Species richness is considered one of the earliest ecological measures of the diversity in a community (Krebs 2014). The indicator is easily understood and can be estimated from most survey data that is based on an accurate effort to identify the species present in a community.

Ecologists differentiate species diversity into three components of alpha, beta, and gamma diversity (Krebs 2014). Alpha (α) diversity is local diversity, such as the diversity of a coral reef zone or a reef. Gamma (γ) diversity refers to the total regional diversity of a large area that contains several communities, such as Western Indian Ocean or the Great Barrier Reef. Beta (β) diversity is a measure of how different community samples are in an area or along a gradient such as distance offshore, depth or latitude from south to north. Beta diversity links alpha and gamma diversity, or local and regional diversity (Whittaker 1972). It is straightforward to estimate alpha and gamma diversity, but the measurement of beta-diversity are controversial (Ellison 2010, Ricotta 2017).

Knowledge of species richness is frequently used in conservation and management decisions. Theoretically more diverse systems, have more species that are likely to fulfill similar ecological functions (Walker 1992), and species differ in sensitivities to disturbances (Elmqvist et al. 2003), therefore higher species richness increases the likelihood of a disturbed community retaining species to perform functional roles and maintain ecological processes.
For this reason it is accepted that higher species richness is more desirable, and species richness to some extent reflects the diversity of ecosystem function performed in a community.

Coral species composition has been reported to change in communities where ecosystem condition has changed or in response to acute disturbance events (McClanahan 2000, McClanahan and Muthiga 2014, 2014, McClanahan et al. 2014). For example a coral reef that is degraded by a bleaching event, may loose coral cover and dominant taxa, such as *Acropora spp*., may decrease in abundance. However, the reduced competition associated with the loss of previously dominant coral species may facilitate the recruitment into the community of less competitive or more rare coral species, that were previously not present, replacing species lost as a result of the disturbance. In this case it is possible that coral communities in different condition have the same number of species and therefore the same species richness (Cheal et al. 2008). Therefore coral species richness alone does not necessarily provide insight into the condition of a coral reef community and alternative approaches have been developed, such as functional assessments and diversity indices that consider ecological interactions, abundance and species richness.

To detect a change or difference in species richness using data from counts of species, assumes that all species are detected, and that the detectability of the different species is the same or at least does not differ between the surveys or communities that are compared. Detectability of a species is the probability of detecting at least one individual of the species in a survey, provided at least one individual of that species is present in the area of interest during the time of the survey (Boulinier et al. 1998). Field surveys rarely detect all of the species that exist in a community (Fisher et al. 2015). Corals are attached to the reef substrate and more easily observed which may facilitate a complete count of species within a defined area. Nonetheless, detection of corals varies because species differ in size and favored microhabitat amongst other life history traits (Veron and Stafford-Smith 2000). Movements of organism, such as fish, insects, invertebrates or birds, further complicate detection often making it impossible to enumerate all of the species in a community (Krebs 2014). Therefore species are not equal in their detectability either.

Overall species richness is most useful for describing the condition of coral community when combined with other information. Useful complementary information includes knowledge of the species present and the relative abundance of species present often captured as species heterogeneity and evenness (Krebs 2014). It is also wise to restrict species richness assessments to specific groups of organisms, in this case corals, because of the differing complexity of assessing all of the organisms in a coral reef community.

## This indicator is relevant to

Coral reef condition
Coral reef resilience

## Benchmarks and thresholds

Coral species richness can be compared to the reported species richness of the Western Indian Ocean (Veron and Stafford-Smith 2000, Obura 2012, Devantier and Turak 2017, Devantier et al. 2020). Comparison should only consider corals that are characteristic of the habitat of interest, given current understandings of coral reef zonation and coral community composition correlated to environmental parameters that include depth, temperature, and wave exposition (Goreau 1959, Chappell 1980, DeVantier et al. 2006, Edmunds and Leichter 2016, Devantier and Turak 2017, Devantier et al. 2020).

One study of scleractinian coral species richness in northern Madagascar, where the total number of species recorded was 292, reported an average of 90 species per site (Standard deviation ± 23), with a range 33–125 amongst all sites (Devantier and Turak 2017). Another study reported average species richness in northern Madagascar as 81 species per site (Standard Error of the mean ± 6.3). Coral species richness in northern Mozambique is slightly lower with a total of ~ 250 species recorded (Obura 2012, Ateweberhan and Sheppard 2018, McClanahan et al. 2018). Both studies define a site to be an area of 0.5 ha.

Therefore to be conservative we suggest:

- 68 species in a coral reef area of 0.5 ha to be an average value for reefs in Northern Mozambique.
- less than 51 (75 % of average) species in a coral reef area of 0.5 ha to reflect a "poor" coral reef condition, and
- less than 34 (50 % of average) species in a coral reef area of 0.5 ha to reflect a "degraded" coral reef.

However, we believe these thresholds are likely to be adapted in response to improved monitoring of coral reefs in Mozambique, and we recommend caution and careful consideration of natural habitat features that may result in lower species richness when applying these suggested thresholds.

## Readiness

Species richness is an easily calculated metric, that can be compared between sites and surveys provided the species at each site are accurately distinguished, but not necessarily identified. However, accurate taxonomic surveys to species level can provide greater insights to coral reef condition.

## Methods and materials

Species richness is the number of different species and is counted in a sample of a community. The sampling effort is described quantitatively with reference to the time (e.g. minutes in a timed swim), the number of replicates used to sample the community (e.g. quadrats, transects), the habitat or the size of the area surveyed.

Coral species richness requires only that species are differentiated and not necessarily identified. Species richness can be estimated by counting the number of unique coral species in quadrats (e.g. 1 m × 1 m and adding the number of new species found in each quadrat (or sampled area) to give a cumulative series of the total number of species observed (Bakus 2007, Krebs 2014). Any standardized search area and time can be used such as belt transects (e.g. 1 m × 20 m), periods of time spent searching (e.g. a 45 minute dive) or an area of reef. However, it is preferable to have multiple separate and random samples.

An important consideration is that ideally the sampling effort upon which species richness is estimated should be standardized (Gardener 2014). The larger the area that is sampled, or the longer species are searched for in a given area, the more species are likely to be identified. When sampling effort differs, methods to standardize species richness such as rarefaction, jackknife estimates or bootstrap procedures that take into account different sampling effort should be applied (Gardener 2014, Krebs 2014)

Species richness can be plotted as a species-accumulation curve, which displays the number of quadrats along the x-axis and the total number of coral species along the y-axis (Krebs 2014). Species-accumulation curves can be used to compare patterns of diversity between communities. The species- accumulation curve also provides an indication of how many samples are required to describe species richness when the curve becomes asymptotic. It is also important to realize that it is unlikely that all of the species present in an area will be identified (Krebs 2014) even if the number of new species identified declines with each new sample. However modern photographic records of a sampled area with improved resolution may allow for a complete count of the coral species in an area (Edwards et al. 2017).

An alternative approach to estimating the coral species richness of a reef is to use statistical software to predict species richness curves based upon random samples. A variety of statistical software packages and packages in the R language have the capacity to estimate species richness (e.g. vegan, SPECIES, rich, SpadeR, BiodiversityR). Rarefaction can be used to adjust for variation in species abundance and sampling effort, and for example in the package vegan this is undertaken with the command rrarefy().

A limiting factor to describing coral species richness is the difficulty in accurately identifying coral species in the field. There are instances when richness is calculated from genera observed in a community instead of species (McClanahan et al. 2012). Although the estimation of richness from genera provides less detailed insight to the diversity of a coral community, it provides the advantage of simplifying the taxonomic skills required for surveys, and as a result increases the reliability of taxonomic identifications.

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# 7. Coral species diversity

## Indicator Name

Coral diversity

## Alternative indicator names

Species diversity, taxonomic diversity, diversity indices, diversity

## Importance

High

## Definition

Coral (or species) diversity of a reef community is a combined measure of the number of species present in an area and the abundance of each species in an area (Gardener 2014). The community is represented by a sample of a specific area and an equal sampling effort, for example an equal numbers quadrats or transects, is required for direct comparison of two communities.

There are many indices that can be used to describe the diversity of a community (Gardener 2014, Krebs 2014), including:

- Simpson's Index
- Shannon entropy
- Effective species
- Evenness
- Berger-Parker dominance index
- Fisher's alpha
- Rényi entropy
- Tsallis entropy

Indices and methods used in studies focused on comparing or assessing the diversity of scleractinian corals include:

- Chao 1 (Devantier et al. 2020)
- Shannon diversity (Díaz-Pérez et al. 2016),
- Pielou´s evenness (Díaz-Pérez et al. 2016),
- taxonomic diversity (Brown et al. 2002, Díaz-Pérez et al. 2016),
- taxonomic distinctness (Brown et al. 2002, Díaz-Pérez et al. 2016)
- Simpson's Index (Brown et al. 2002)
- Dendrograms of affinity the Dice coefficient (Veron 2015)
- Predicted maximum species richnes (Smax) (Obura 2012)
- Cluster analysis (ANOSIM) (Obura 2012)

## Background and Reasoning for choosing the indicator in Mozambique

Theoretically more diverse systems, have more species that are likely to fulfill similar ecological functions (Walker 1992), and these species have different sensitivities to disturbances (Elmqvist et al. 2003) increasing the likelihood of a disturbed community retaining species to perform each functional role.

Experimental and theoretical studies support that ecological communities with greater species richness and diversity are more resilient to disturbances and more likely to retain or regain ecological functioning. Species may decline to a point where they no longer effectively perform an ecosystem function after a disturbances, however the functional roles undertaken by the species that decline are more likely to be compensated for by other species that are more tolerant of the disturbance, and which perform similar ecological functions (Loreau et al. 2001, Elmqvist et al. 2003, Bellwood et al. 2004).

Species diversity indices attempt to take into account the number of species present in a community as well as the relative abundance of species in the community. Among the most commonly used species diversity indices are the Shannon and Simpson's indices (Gardener 2014). Two coral reef communities or sites may have the same species richness (number of species) and the same species composition, however the relative abundance of the species present can be markedly different and this can be reflected by a diversity indices.

Species diversity índices when considered alone do not necessarily provide insight to the condition or state of degradation of a coral reef community (Cheal et al. 2008). For example a stable and resilient coral reef community may have the same estimated diversity but be characterized by species that represent a coral reef in good condition (e.g. numerous long lived species from the coral genus *Porites*). In contrast, a degraded coral reef community may have the same number of coral species in similar abundances but the species present could be opportunistic and representative of a reef in lesser condition (e.g. species from the genus *Pocillopora*, young individuals of species from the genus *Acropora*). These different coral reef communities might have identical indices of diversity.

## This indicator is relevant to

Coral reef condition
Community diversity

## Benchmarks and thresholds

The diversity of a coral community can be compared to other reefs the Western Indian Ocean region, particularly areas that are in good condition as a result of remoteness or effective marine protected areas, however this is challenging given there are few published reports and may require specific surveys.

DeVantier et al. (2020) list a Chao 1 indices value of 280 for Northern Mozambique, where they report total species richness to be 239, with an average of 81 (Standard Error of the Mean ±6.3) for sites of 0.5 ha area.

## Readiness

Coral diversity is not a readily comparable indicator, as it requires data for both the species richness and abundance of each species. However it is within reach of basic monitoring efforts required to describe other key aspects of a coral community and compare reef environments, such as coral cover. It is important to realize that each indices is unique and these cannot be compared directly.

## Methods and materials

Basic survey techniques are required to describe a coral community which may range from timed searches to more quantitative use of transects and quadrats (English et al. 1997, Hill and Wilkinson 2004). A diversity index in then calculated usually with knowledge of the number of unique species present and their abundance (Gardener 2014, Krebs 2014).

Most species diversity indices can be calculated using an MsExcel, Numbers or a similar mathematical spreadsheet software, however numerous packages in R also facilitate the calculation of species diversity indices (e.g vegan, BiodiversityR) (Gardener 2014).

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# 8. CORAL POPULATION SIZE-FREQUENCY DISTRIBUTION

## Indicator Name

Coral population size-frequency distribution

## Alternative indicator names

coral population size distribution, coral population size structure, coral size class frequency, coral size class distribution, coral size-frequency distribution

## Importance

High

## Definition

The size-frequency distribution of a coral population is the description of corals in that population according to the frequency of colonies in defined size classes. The distribution of corals in different size classes, and the change in this distribution over time, gives insight into the demography and dynamics of a coral population (e.g. rates of recruitment, growth reproduction and mortality). This information is useful in assessing the condition of a coral population, or community when the populations of several species are assessed.

## Background and Reasoning for choosing the indicator

Size class frequency data provides insight to the demographics and the dynamics of a coral community. Information on coral populations, such as the spatial and temporal variation in patterns of recruitment, partial mortality, and longevity among and within coral species, can be interpreted from the size-structure of populations (Bak and Meesters 1998, Bar and Meesters 1999, Meesters et al. 2001, Crabbe 2009, Madin et al. 2014). This information provides insight into the potential for a coral community to recover from, or to resist, disturbance as well as the overall resilience of a coral reef (Mumby and Steneck 2008, McClanahan et al. 2012, Obura et al. 2018).

The size of coral alone can be misleading in the estimation of age because most corals have a clonal life-form and experience fragmentation, fission, fusion, and partial mortality which all affect size without affecting age (Hughes and Jackson 1980, 1985, Babcock 1991). However, many life-history processes are strongly size-dependent for corals, such as when colonies are reproductive, fecundity, potential reproductive output, growth rates and the likelihood of survival or partial mortality (Harrison and Wallace 1990, Hughes and Tanner 2000, Madin et al. 2014). Therefore, describing the size-frequency distribution is a useful approach to studying the demographics of a coral population, predicting future trends and interpreting past events (Connell et al. 1997, Edmunds and Elahi 2007)

Coral growth rates vary between colony size, taxa, habitat and region (Harrison and Wallace 1990, Pratchett et al. 2015), however knowledge of local growth rates can be used to gain insight into the frequency of disturbances, and the time passed since disturbance events (Connell et al. 1997, 2004, Gilmour et al. 2013). For example a cohort of corals with a diameter of 30 cm from a species that extends its skeleton by ~1 cm per year (e.g. diameter increases by ~2 cm per year) are likely to have taken 15 years to reach that size, which suggests they recruited approximately 15 years earlier. Bimodal distributions, or a relatively high prevalence of two average sizes such as 30 cm and 60 cm may indicate a cohort that survived a disturbance and continued to grow to 60 cm with an smaller cohort that recruited following the disturbance having reached an average size of 30 cm, and also suggests that major disturbances impacted the coral population at 15 year intervals. Rarely is data for the size-frequency distribution of coral populations so simplistic (Connell et al. 1997, 2004), but a clear examples of size related dynamics of coral cohorts and surviving colonies is supported by observations of corals recovering from mass bleaching events (Gilmour et al. 2013).

The life history characteristics of corals are unique to each species (Darling et al. 2012, Madin et al. 2016c, 2016a). For this reason size-frequency distributions are most useful when collected for individual coral populations within a community. Practicality often limits the ability to describe the size-frequency distribution of all of the coral species present in a reef community, either because of the time required, or in the case of less abundant corals, because the numbers of individuals present in an area is insufficient to describe a population. However, it is possible to select corals that are representative of different strategies in the community, such as slower growing and longer living *Porites spp.* and faster growing and shorter living *Acropora spp.*. The two genera also have greater influence than other coral genera on other coral reef organisms such as fish and invertebrates (Coker et al. 2014, Pratchett et al. 2018).

Coral reef condition can be assessed from the eveness or skewness of sizes of corals in a population, which may reflect growth rates, recruitment failures or success, and isolated mortality events or survival trends. The degree and direction of skewness can be species specific, as a result of life history traits that influence reproduction, growth and survival (Harrison and Wallace 1990, Darling et al. 2012, Anderson and Pratchett 2014, Madin et al. 2016b). Positively skewed coral populations, with high numbers of recruits and decreasing numbers of progressively large colonies are characteristic of most locations (Hughes & Jackson 1985, Meesters et al 2001, Anderson & Pratchett 2014).
A negatively skewed coral population, with higher frequency of large colonies and fewer juveniles, suggests poor recruitment or high mortality of smaller colonies, and may indicate a population is at risk of declining (Guzner et al., 2007; Alvarado-Chacon and Acosta, 2009; Madin et al., 2014). A wide range of size classes is expected of a coral population in good condition (Bak and Meesters, 1998; Meesters et al., 2001). Multiple size specific peaks and lows in the abundance colonies can provide insights into past disturbance events that affected specific cohorts of coral recruits or populations as a whole (Gilmour et al. 2013, Pisapia et al. 2019). Examining the size-structure of coral populations

not only provides insight to past events but also enables predictions of future changes in coral populations (Edmunds and Elahi 2007).

## This indicator is relevant to

Coral community resilience
Coral community recovery from and resistance to disturbance events,
Coral demographics,
Temporal dynamics of a coral population or community

## Benchmarks and thresholds

The size-frequency distributions of a coral population in good state, with unhindered reproduction and recruitment and that has not suffered acute mortality or partial mortality as a result of disturbance events is likely to be positively skewed, having a high number of small corals and progressively less corals as colony size increases.

A negatively skewed coral population (i.e. with few recruits and increasing more larger corals) is likely to be recruitment limited suggesting there is a deleterious influence on fecundity of corals (e.g. pollution) or the survival of recruits (e.g. habitat degradation with sediments and algal turfs).

Historical knowledge of local coral population size-frequency distributions for key coral populations (e.g. *Acropora spp*., *Porites spp*.) or of those from similar coral communities can help to assess the condition of a coral community and identify or predict the likelihood of a coral reef community persisting at least in its current state under local environmental conditions and stresses. Changes in the size-frequency distribution of a coral population over time will also provide insight into the dynamics and condition of a coral reef community.

## Readiness

Size class data for coral communities is not frequently collected, which limits regional comparisons. However, reference to published studies can enable comparisons that provide insight to the condition of a coral community. Size-frequency data is most useful when collected at regular time intervals at a site, such as each year, to accurately describe temporal changes in the demographics of coral populations or a community. The collection of size class data requires observers with experience in coral observation. Corals need to be identified, ideally to species level. Partial mortality and identification of individual colonies may also confuse an inexperienced observer and result in underestimated colony sizes or incorrect colony counts (Hughes & Jackson 1980, 1985).

## Methods and materials

Corals within a specified search area are identified and their size measured so that they can be allocated to size classes. Ideally corals are distinguished by species because life history traits such as growth rate, partial mortality, fragmentation and size differ between taxa (Harrison and Wallace 1990, Madin et al. 2014, Pratchett et al. 2015).

The search area is frequently defined by a belt transect or a quadrat, which can be replicated at a site (English et al. 1997, Hill and Wilkinson 2004). The size and number of the belt transects is often in part determined by the time available for an observer using SCUBA during a single dive. Modern approaches to create high resolution photo-mosaics can facilitate surveys of larger area by using in water time to gather images and allow for observations to be made in a laboratory setting with less pressing time restrictions (Edwards et al. 2017). The size of a search area is best determined when consideration is given to the taxa of interest and their local abundance. For example larger belt transects (e.g. 50 m × 5m) may be appropriate for surveying corals of the genus *Porites* with a massive morphology which can be several meters in diameter. Whilst, smaller belt transects (e.g. 10 m × 1m) may be appropriate for species of the genus *Acropora* with digitate morphologies and maximum diameters up to 1 m. The size class distribution of a coral population is described by the frequency with which each size class is observed and therefore determined from the number of colonies observed, and accuracy is improved with increasing numbers of observations. Standardizing the search area is not essential, however this ensures the search effort is consistent and also allows for colony abundance to also be estimated without extra survey efforts, therefore it is highly recommended.

So that the probability of including corals of different sizes is equal, it is important to make observations on the corals that have their geometric centers inside the search area (Zvuloni et al. 2008). Selecting corals that are partially inside the search area would increase the likelihood of including larger corals and including corals that are mostly inside the search area would increase the likelihood of including smaller corals.

The ideal size range for each size class may be a function of the population being studied. Greatest detail is achieved with smaller size classes (e.g. 1 cm bins), which can later be grouped into larger size ranges (e.g. 5 cm or 10 cm bins). However, greater detail may require more time for observation to be made and may require greater replication to show general patterns.

The most accurate observations are made by measuring individual corals and later allocating these to size bins. Most frequently it is the maximum diameter of a colony that is measured, although alternatives include combined measurements of the maximum diameter and the diameter perpendicular to this, the two dimensional area of a colony as seen from above (which can be estimated from photographs), or estimates of the coral volume. Measurements can be made with a tape measure or calipers in situ, or from calibrated images using dedicated software (e.g. CPCe, ImajeJ, Photoshop). Measurements from photographs require two-dimensional scales to be included in the image, and measurements from stereo-photography require prior calibration of cameras with appropriate measurement software and calibration devices.

It is also possible to predetermine size ranges and allocate coral colonies to these without accurately measuring each colony. This may reduce the time required in the water for a survey or to gather data from images. However, pre-determined size classes reduce resolution of the data and the ability to redefine size classes, which might restrict the ability to compare results to another study, between years, or between observers when.

Photogrametry is also a highly accurate approach to recording the size for corals (e.g. TransectMeasure www.seagis.com.au). Accurate measurement of the size of corals (and other benthic organisms) from photographs is more time consuming and costly. There are also equipment requirements (e.g. cameras, calibration hardware, software licenses). However, the approach may extend the depth of surveys if drop cameras or automated underwater vehicles are used, and can reduce the risk of observer bias. This may also reduce the in water time required to survey habitats allowing for faster fieldwork.

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# 9. CORAL GROWTH RATE

### Indicator Name

Coral growth rate

### Alternative indicator names

Coral calcification rate

### Importance

High

### Definition

Coral growth rate can broadly be defined as the amount of change in the physical dimensions, of a discrete coral colony expressed against a unit of time (e.g. cm/year).

How coral growth is measured differs between studies and coral taxa, largely as a result of biological traits such as colony morphology and skeletal density (Pratchett et al. 2015). Growth may therefore be measured as the change in linear dimensions, planar area, surface area, volume, or mass of the skeleton of a coral colony (Hughes and Jackson 1985, Morgan and Kench 2012, Pratchett et al. 2015). Growth is most commonly measured as linear extension, a unidirectional change in branch length or colony radius, but this may limit comparison between corals with different growth forms and calcification traits, and the most comparable measure of growth is annual calcification or change in weight normalized to a measure of size (Pratchett et al. 2015).

### Background and Reasoning for choosing the indicator in Mozambique

The growth of coral colonies creates calcium carbonate skeletons that make a key contribution to building coral reefs (Goreau 1961, Stearn et al. 1977). Coral growth rates provide insight into the condition of a coral reef community because a reef that erodes faster than it is built will progressively cease to exist (Scoffin 1992). Reef accretion rates must also match rates of sea level rise for croal reefs to persist under conditions of predicted climate change (Perry et al. 2011, 2018). Coral growth rates can also provide insight to sub-lethal stresses but this may require observations over relatively long timeframes of years to decades to record. For example, growth rates of massive *Porites* colonies declined approximately 11% between 1990 and 2005 on the GBR, an indication that the condition of this population of corals and the reef in general has declined (De'ath et al. 2009, 2012).

Coral growth rates respond to light and water quality (e.g. turbidity), temperature, inorganic nutrient levels, heterotrophic intake and aragonite saturation state all of which contribute to spatial and temporal variation in coral growth. Corals vary their investment in growth seasonally in response to light, temperature and chemistry (Edmunds and Putnam 2020). The investment in growth also differs throughout a single colony, with higher growth rates in vertical directions in massive *Porites spp.* (Lough and Barnes 2000) or at the extremities of branches in *Acropora spp*. (Wallace 1999) presumably because at this location colonies receive more light.

Growth rates differ between coral taxa as a result of species-specific life history traits. The direction of growth can be in a singular direction, for example when branches extend in colonies of *Acropora spp.* or can be three dimensional for massive *Porites spp* (Pratcehtt et al 2015). Species-specific investment patterns in skeletal growth may result in high-density coral skeleton which is slower to grow (e.g. *Porites spp*.) or lighter porous skeleton that grows faster (e.g. *Acropora spp*.). Patterns in the investment in growth and growth rates are generalized for coral morphology (Hughes & Jackson 1985, Pratchett et al 2015). Branch extension is prevalent in arborescent colonies and radial extension in tabular morphologies. Whilst, vertical extension is greater in columnar and massive colonies, and encrusting colonies tend to extend horizontally. Historically studies have averaged coral growth rates over time (e.g. years), and multiple colonies, to obtain average growth rates for coral taxa (Buddemeier & Kinzie 1976, Hubbard & Scaturo 1985). This resulted in corals being generalized as “fast growing” or “slow growing” species, often associated with morphology (Pratchett et al. 2015).

Direct measurement of coral growth are undertaken by repeating observations over time of the linear dimensions, area, volume or weight of a colony to estimate averages in growth over time. Detecting change in growth rates usually will require observations taken years to decades apart (Edmunds 2007, Bak et al 2009, Anderson et al 2014). A fixed point of reference in a colony is required against which to measure growth and this can be a tag or a stain or even a distinct point of the colony in a repeatedly taken photograph (Pratchett et al. 2015).

Stains used as a reference to measure coral growth include alizarin, calcein or oxytetracycline and these methods are reviewed by Holcomb et al. (2013). Skeleton that grows during the time period after the coral was stained can be measured as growth. Staining is a destructive measure that requires collection of the coral skeleton for observation in the laboratory, so growth might be measured on a colony only once, or multiple subsamples must be taken from the same colony to measure growth more than once (Morgan & Kench 2012).

Growth can also be measured retrospectively in species of corals that develop growth bands, distinct bands of aragonite that preserve the seasonal growth variation and relate to time (Pratchett et al. 2015). These are mostly observed in coral with massive morphology and are visible as light and dark bands that record annual variation in growth (Lough and Cooper 2011). Modern measurements of these bands use X-radiography, computerized tomography or gamma ray to distinguish the density of coral skeletons, which varies seasonally and thus allows the identification of yearly growth patterns (Lough and Cantin

2014). Observing growth bands also requires destructive collection of cores or cross sections of corals, and care should be taken to patch lesions on a coral colony to minimize partial mortality or vulnerability to pathogens and parasites.

The most common metric of growth is linear extension measured as a unidirectional change in branch length or colony radius (Pratchett et al 2015). However, linear extension may be difficult to compare across taxa and growth forms (Browne 2012), and the most comparable measure of growth across a range of coral growth morphologies is annual calcification or change in weight normalized to a measure of size (Pratchett et al 2015).

### This indicator is relevant to

Sub-lethal stresses to corals
Coral community condition
Coral community resilience
Coral reef accretion

### Benchmarks and thresholds

Reference values for coral growth rates can be measured at local control reefs and at the same reef at different points in time.

General growth rates are reported for a large number of coral species (Lough and Cantin 2014, Madin et al. 2014, Pratchett et al. 2015) and these reported values can provide general reference values.

### Readiness

Low.

Measurements require dedicated surveys and changes in growth rates may only be detectable over timeframes of years to decades. Event though measurements can be made on scale of days.

### Methods and materials

We do not recommend the use of destructive observations of coral growth. The least intrusive approach to measuring coral growth rates is to make direct observations in the corals natural environment at repeated intervals.

We suggest that a small number of representative species, that are widely distributed in the region, are chosen to represent a coral community when using growth rate as an indicator of the condition of a coral reef community.

Growth rates vary throughout a colony so it is important to either be consistent with where growth is measured, or to take multiple samples from a colony (Lough & Barnes 2000, Morgan & Kench 2012). In either case it is important to accurately document where and how growth is measured (Pratchett et al. 2015).

Replication should also ensure that multiple colonies of each species are measured in a survey.

Photographs combined with dedicated image analysis software (e.g. ImageJ, CPCe, EventMeasure) can be used to measure changes in coral area or dimensions and minimize impacts to corals. These approaches are particularly useful for estimating changes in planar area and linear extension (Pratchett et al. 2015), thus are more appropriate for corals with planar growth forms (e.g. encrusting, tabular and foliose corals) or branching corals. It is important to include a two-dimensional scale, maintain perpendicular camera lens angle when taking a picture and to use the same camera where possible or account for camera specific biases.

Photogrammetry measurements can be used for more precise measurements to construct 3D models of coral colonies. Suggested software for these purposes include EventMeasure (www.seagis.com.au) and Agisoft Metashape (www.agisoft.com). Precise volume calculations of colonies modeled at different points in time are made possible by the photogrammetric 3D model approach. The disadvantages include higher costs for equipment and software, as well as more time required underwater, however photogrammetry is a precise and non-intrusive method for measuring coral growth. Not yet widespread, the technology for these approaches is advancing rapidly and benefiting from increasingly available technologies and software that allow for more efficient workflow (Burns et al. 2015, 2016, Raoult et al. 2016, House et al. 2018).

Buoyant weight measurements are an alternative, accurate and relatively non-intrusive method to measure growth, even on timescales of days (Jokiel et al 1978, Spencer Davies 1989). However, the method does require coral colonies, or fragments, that are weighed to be detached from the reef substrate.

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# 10. STRESS-TOLERANT CORAL TAXA

## Indicator Name

Stress-tolerant coral taxa

## Alternative indicator names

Disturbance resistant corals

## Importance

High

## Definition

The presence of coral taxa that are tolerant of a stress or disturbance affords a coral reef community greater resilience.

## Background and Reasoning for choosing the indicator in Mozambique

Coral species differ in their phenotypes and life history traits, which results in a range of susceptibilities to disturbances and stressors (Darling et al. 2012, Madin et al. 2014, Zawada et al. 2019). When there is higher the prevalence of coral taxa that are resistant to, and capable of recovering from, a specific disturbance the greater, then a coral reef community is likely to be more resilient (McClanahan et al. 2012). The phenotypic traits of scleractinian corals that are predicted to increase the contribution a species makes to a coral community include low partial mortality, high wound healing rate, high skeletal growth rate, bleaching resilience, infectious disease resilience, and high sexual reproductive output (Baums et al. 2019). Corals that are characteristic of deep water habitats (>100 m), have widespread or cosmopolitan distributions, are non-symbiotic, have a solitary nature or small colony size and that show bleaching-resistant traits have more successfully survived disturbances since the Triassic, and it is suggested that modern scleractinian corals with these same traits have relatively stable populations, while those lacking them are presently decreasing in abundance and diversity (Dishon et al. 2020).

The majority of reef building scleractinian corals have symbiotic zooxanthellae and are colonial (Veron and Stafford-Smith 2000, Veron 2015, Madin et al. 2016, Hoeksema and Cairns 2018). However, reef building scleractinian corals are known to differ in their susceptibility to specific stressors. Examples of stressors for which different levels of resistance has been described include physical damage from storms and waves (Harmelin-Vivien 1994, Zinke et al. 2018, Bozec et al. 2020), bleaching events (Marshall and Baird 2000a, Loya et al. 2001, McClanahan et al. 2007, van Woesik et al. 2011), sedimentation (Nugues and Roberts 2003, Vermeij and Bak 2003, Sofonia and Anthony 2008), or predations by crown-of-thorns starfish (Taylor and Ormond 1976, Pratchett 2001, Daud and Tokeshi 2011, Kayal and Kayal 2017, Keesing et al. 2019). Therefore it is most useful to discuss likely stressors or disturbances individually and to identify the coral taxa with greater resistance within a specific context.

*Storms and waves*

Corals exist in a range of habitats which present a variety of potential disturbance levels from wave action and storms, and coral morphologies and skeletal densities have adapted to the risk of physical damage. Sturdy hemispherical, compact and short branching or encrusting morphologies are characteristic of shallow reef flats and crests where wave energy is high (Dollar 1982, Dollar and Tribble 1993, Veron and Stafford-Smith 2000). Cyclones and storms generally damage and break corals that are shallower than 20 m, with branching and tabular coral morphologies more susceptible to breakage than upright massive corals (Scoffin 1993). Large massive colonies, such as massive Porites spp., are more likely to survive cyclones and over time may become abundant in habitats that are generally sheltered from waves as a result of successive storm events (Fabricius et al. 2008).

*Bleaching*

Corals of upright finely branching morphologies (e.g. *Acropora spp*. and *Pocillopora spp*.) have historically bleached more than corals with massive morphologies (Marshall and Baird 2000b, Loya et al. 2001, McClanahan 2004, Mcclanahan et al. 2007).

The bleaching stress to corals as a result of temperature anomalies may be aggravated by increases in light intensity (Hoegh-Guldberg 1999, Rosic et al. 2020). Morphologies that self shade, or skeletal traits that reduce light capture, may alleviate this additional stress and facilitate partial survival of colonies during bleaching events (Titlyanov 1991, Coelho et al. 2017, Enríquez et al. 2017).

The resistance to bleaching by the symbiotic zooxanthellae characteristic of each coral taxa is also key to determining the relative susceptibility of corals to bleaching (Baker et al. 2008). Changes in relative proportions of coral colonies with resistant zooxanthellae symbionts, may have resulted from greater removal of sensitive zooxanthellae during bleaching events, increasing the prevalence of more heat-resistant coral-zoxanthellae associations altering the relative susceptibilities of coral taxa to bleaching (Guest et al. 2012, Baker et al. 2020, McClanahan 2020).

*Sedimentation*

Corals are reported to become more resistant to sedimentation as colony size and corallite diameter and tissue thickness increase (Rogers, 1990; Stafford-Smith, 1993; Fabricius, 2005, Erftemeijer et al 2012). Colony growth forms that are more upright also shed sediments more easily than corals that are more horizontal (Erftemeijer et al 2012).

*Crown-of-thorns starfish*

Crown-of-thorns starfish are reported to preferentially feed on *Acropora spp.* and *Pocillopora spp.* with lower interest for massive colonies such as *Porites spp*. and

*Favites spp. Favia spp*. (Taylor and Ormond 1976, De'ath and Moran 1998, Pratchett 2007, Keesing et al. 2019). Whilst, corals of the genus *Pocillopora* are often less prone to COTS predation than *Acropora spp* because they are defended by symbiotic crabs that live on the coral (Pratchett 2001). The presence of stinging hydrozoan Milleporidae colonies can also reduce predation of corals by creating a barrier to COTS (Kayal and Kayal 2017)

### This indicator is relevant to

Coral reef resilience
Coral community condition
Resistance to disturbance events and stress
Coral reef zonation

### Benchmarks and thresholds

Specific benchmarks and thresholds are not available for the proportion of stress-tolerant taxa in a coral reef community and to our knowledge comparisons are qualitative or related to the relative number or abundance of specific taxa.

### Readiness

An appraisal of the presence of stress resilient taxa in a coral community is possible from survey data that identifies the coral species or genera present and their abundance in a community. This should be complemented with a review of the scientific literature that reports tolerance and susceptibilities of coral taxa to stress, which in turn should be verified locally.

Application requires a reasonable degree of knowledge of how specific stressors or disturbance events impact a coral community, as well as field survey experience and identification skills.

Comparison with other reefs and regions can be undertaken qualitatively and based on community composition or the number of stress-tolerant taxa present.

### Methods and materials

To assess the condition of a coral reef community with reference to the prevalence of stress-tolerant taxa requires:

- Quantification of the abundance, benthic cover and ideally size-frequency structure of individual coral taxa, preferentially to species level (alternatively identified or grouped to the level of genus)
- Description of species richness
- Evaluation of the strength and frequency of specific disturbances and/or stresses

The abundance of stress-tolerant corals otherwise relies on similar survey methods as used to assess coral cover and diversity

Corals should be surveyed quantitatively and widely used methods include line intercept transects, point intercept transects, and quadrats (based on points or area) (English et al. 1997, Hill and Wilkinson 2004).

Photoquadrats are frequently used to describe coral communities.  Photos can either be taken by a person in the water or using an automated or remotely controlled approach such as a drop camera or a camera mounted to an unmanned underwater vehicle (UUV) or autonomous underwater vehicle (AUV) (Bryant et al. 2017). Image quality usually limits photoquadrats to the areas of the order of 1 $m^2$ or smaller, unless multiple images are joined. To quantify coral cover from photographs, replicated points or areas of the image are described using software such as Coral Point Count with MsExcel extensions (CPCe) (Kohler and Gill 2006)or PhotoQuad (Trygonis and Sini 2012), points or subdivisions overlayed to an image using printed photographs or image software (e.g Photoshop), or  semi-automated identification platforms such as CoralNet (Beijbom et al. 2012).

The advantages of technological developments and the lower cost and greater availability of high quality images of the benthic community include collection of data for a variety of metrics (percentage cover, species composition, disease or bleaching incidence), that can be undertaken using point counts, digitized polygons, area subdivisions or other methods used to analyze underwater imagery (Kohler and Gill 2006, Beijbom et al. 2012). Still photography is generally preferred, however video transects have also been used (Hill & Wilkinson 2004).

Photo-mosaics and 3D maps of coral reef substrate are another approach for capturing detailed images of larger scale, from which coral cover can be estimated along with other information for a benthic community (Lirman et al. 2007, Edwards et al. 2017). The technology for these approaches is advancing rapidly and benefiting from increasingly available technologies and software that allow for more efficient workflow (Burns et al. 2015, 2016, Raoult et al. 2016, House et al. 2018)

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## 11. CORAL RECRUIT ABUNDANCE

### Indicator Name

Coral recruit abundance

### Alternative indicator names

Coral recruit density

### Importance

Critical

### Definition

Coral recruit abundance is defined here as the number of corals, of maximum diameter no greater than 5 cm in diameter, within an area of coral reef substrate. This definition is consistent with numerous field based studies that focus on surveying coral recruits that are visible to the naked eye on the reef substrate (usually larger 1 cm in maximum diameter) (Hughes et al. 2007, Jonkers et al. 2008).

Coral recruits visible to the naked eye can be surveyed in the reef environment and are representative of the outcomes of natural processes acting on the coral reef community. This is different to studies of corals settled onto artificial surfaces which are indicative of the supply of larvae and planulae to a habitat (Mundy 2000), but may not fully represent the influence of the physical, chemical or biological environment of the reef substrate on the selection of a surface to settle on, or the resulting consequences for post settlement survival.

### Background and Reasoning for choosing the indicator

Coral recruitment is a key demographic process that contributes to the ongoing replenishment and the recovery from disturbance of a coral community (Connell et al. 1997, Hughes and Tanner 2000, Lukoschek et al. 2013). The abundance of coral recruits and the rates at which new coral recruits enter a coral community are frequently assessed to gain insight to the condition, resistance and resilience of a coral reef community (Hughes et al. 2007, Fabricius et al. 2017).

Insight is gained to the relative condition of reefs, because coral recruit abundance is spatially variable between reefs and habitats as a result of the numerous factors that influence the success of coral recruitment (Edmunds et al. 2010, 2015, Jonker et al. 2019).

The net outcome of coral recruitment is the result of the combined success in several steps in the life cycle of a coral:

1) The abundance and fecundity of coral species contributing to coral recruitment, their individual reproductive strategies and life history traits, all

of which contribute to variation in the outcomes coral recruitment (Harrison and Wallace 1990, Baird et al. 2009, Álvarez-Noriega et al. 2016).

2) The supply of coral larvae and planulae to settle on the reef substrate, which is in turn a result of the fecundity of existing coral communities, the connectivity of a habitat with sources of larvae and planulae, the longevity and survival of coral larvae and planulae (Harrison and Wallace 1990, Hughes et al. 2000, Ritson-Williams et al. 2009, 2016)
3) The settlement of larvae and planulae on the substrate which is influenced by biological interactions, physical and chemical characteristics of the environment and the fitness of the larvae or planulae (Harrison and Wallace 1990, Maida et al. 1995, Ritson-Williams et al. 2009, 2016, Arnold et al. 2010, Arnold and Steneck 2011, Webster et al. 2015, Speare et al. 2019).
4) The early survival of corals, which is a result of the stresses and disturbances in the physical environment (e.g. sedimentation, temperature anomalies), biological interactions (e.g. competition, predation) (Vermeij 2006, Box and Mumby 2007, Mumby 2009, Arnold and Steneck 2011, Webster et al. 2015, Moeller et al. 2017).

### This indicator is relevant to

Coral community recovery
Coral reef resilience
Coral reef condition

### Benchmarks and thresholds

We do not believe it is possible to define a threshold for the abundance of coral recruits in a coral reef community because this is a result of the available habitat for recruits which is impacted by the space occupied by established coral colonies or the coral reef complexity which determines surface area of reef substratum. However, it is possible to compare the recruit abundance at a reef through time and describe increases or decreases in coral recruitment relative to the available habitat.

### Readiness

The abundance of coral recruits is a widely reported metric and can easily be measured with basic resources and training.

### Methods and materials

The methods that can be used for surveying coral recruits are similar to those used for corals and other benthic organism in general, although they must be adapted to take account of the smaller size of coral recruits. Coral recruits are relatively small, often cryptic, and therefore should be searched for in smaller areas than mature corals, as more time will be required for accurate surveys of a similar area.

Belt transects (e.g. 5 m × 0.5 m to 10.0 m × 1.0 m) or quadrats (e.g. 0.5 m × 0.5 m to 1.0 m × 1.0 m ) are widely used methods to quantify coral recruit abundance in situ and should be replicated at a site (Hughes and Tanner 2000, Jonkers et al. 2008, Gilmour et al. 2013). Observers should be familiar with the identification of coral genera and able to apply this to identify clearly distinguishable coral recruits (Jonkers et al. 2008, Jonker et al. 2019). Photographs and photo-mosaics may also been used to detect coral recruits or to make a record of the individual recruits (Hughes et al. 2007, Jonkers et al. 2008, Edwards et al. 2017b). However, the two-dimensional nature of photographs may result in a greater proportion of coral recruits not being detected when relying on photographs compared to the naked eye, especially in complex habitats and communities because of the preferential settlement of corals in cryptic habitats (Hughes et al. 1999, Arnold et al. 2010).

Improving image technologies may improve photographic surveys of coral recruits. Fluorescence and ultra-violet lighting may facilitate the photographic detection of coral recruits but require more costly and delicate equipment, or nocturnal surveys (Baird et al. 2006, Schmidt-roach et al. 2008, Palmas et al. 2014). Images may also be combined to map or model reef substrate and accommodate 3D surface features (Lirman et al. 2007, Burns et al. 2015, 2016, Raoult et al. 2016, Edwards et al. 2017a, House et al. 2018)

Coral recruitment will normally be expressed as the number of recruits per meter square (n/m$^2$). Coral recruitment can also be assessed as a rate if the abundance of coral recruits is quantified per unit of time (e.g. year). Repeated observations of permanent plots over time can be useful to identify new recruits at each survey time, survival and growth rates (Connell et al. 1997, 2004, Hughes and Tanner 2000, Hughes et al. 2007). The accuracy of repeated surveys is increased by mapping the location of recruits in plots to enable the fate of each recruit to be followed. Photographic records of individual recruits will provide insight into the interactions and microhabitat that influences recruit survival. Photogrammetry, may be used to increase the accuracy of measurements made to monitor changes in the size of coral recruits.

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# 12. CORAL RECRUIT SURVIVORSHIP

## Indicator Name

Coral recruit survivorship

## Alternative indicator names

Coral recruit mortality

## Importance

High

## Definition

Coral recruit survivorship is the proportion of coral recruits that survive from one year to the next. A coral recruit is defined as a coral with maximum diameter of less than 5 cm.

## Background and Reasoning for choosing the indicator in Mozambique

Describing temporal trends in the abundance and survival of coral recruits will provide insight into demographic processes of coral communities, such as recruitment, growth, and mortality (Hughes and Jackson 1985, Bak and Meesters 1998, Hughes and Tanner 2000a, Hughes et al. 2007). Demographic processes determine the abundance of corals in a population or community, and together with knowledge of the conditions at a reef, and how these condition impact corals, this knowledge allows for predictions and interpretations of trends in the condition of a coral reef community (Hughes and Tanner 2000b).

Following the survivorship of a coral recruit community is likely to provide insight into impacts of specific disturbance events and stressors. Examples include impacts to the survival of coral recruits resulting from cyclones (Hughes 1994, Connell et al. 1997, 2004), bleaching events (Depczynski et al. 2013), biological interactions including competition with macroalgae (Hughes et al. 2007) and fish or echinoderm grazing (Mumby 2006, 2009, Mumby et al. 2007, Leary et al. 2013).

Detailed understanding of the recruitment processes of a coral reef provides greater ability to understand the resilience and capacity of a coral community to recover from disturbances (Hughes 1994, Connell et al. 1997, Mumby and Steneck 2008, McClanahan et al. 2012), therefore assessing coral recruit survival rates, and factors that influence this, is a useful way to assess the overall condition of a coral community and coral reef.

### This indicator is relevant to

Coral community recovery
Coral community resilience

### Benchmarks and thresholds

We are unaware of regionally defined thresholds for the survivorship of coral recruits necessary to sustain a coral community. However, local assessment of the survival of coral recruits, together with growth rates, numbers of corals that reach a reproductive size, coral size-frequency distributions, and size-specific mortality will provide insight to the levels of recruitment required to sustain a coral reef community. Interpretations will largely benefit from modeling studies (Hughes and Tanner 2000b, Wolff et al. 2018, Mcleod et al. 2019). It may also be possible to set reference levels by comparing recruit survivorship to local control reefs that are perceived to be in good condition and resilient.

### Readiness

Survivorship is not a readily available metric for comparison. Although the abundance of coral recruits is a widely reported metric, temporal trends in survivorship require detailed observations of the same corals at long-term study sites.

### Methods and materials

Similar methods are used as for assessing coral recruit abundance, with the added requirements of making repeated observations on the same corals at long-term study sites. Ideally coral recruit survivorship should be followed through events of key interest, such as cyclones, rainy seasons which result in high sedimentation in reef waters or bleaching events as well as periods free from disturbance, therefore detailed local understanding of recruit survivorship may only be possible after multiple yearly surveys.

Repeated observations of permanent plots over time are required to identify new recruits at each survey time, survival and growth rates (Connell et al. 1997, 2004, Hughes and Tanner 2000b, Hughes et al. 2007). The accuracy of repeated surveys is increased by mapping the location of recruits in plots to enable the fate of each recruit to be followed. Photographic records of individual recruits will provide insight into the interactions and microhabitat that influence their survival, and also increase accuracy of following individual recruits through time.

The shorter the time interval between observations the greater the insight into processes impacting coral recruit survival. Therefore it may initially be advisable to monitor coral recruits at monthly intervals (Hughes et al. 2007), however yearly surveys may also provide considerable insight (Hughes and Tanner 2000b).

Belt transects (e.g. 5 m × 0.5 m to 10.0 m × 1.0 m) or quadrats (e.g. 0.5 m × 0.5 m to 1.0 m × 1.0 m) are widely used methods to quantify coral recruit abundance in situ and should be replicated at a site (Hughes and Tanner 2000b, Jonkers et al. 2008, Gilmour et al. 2013). Observers should be familiar with the identification of coral genera and able to apply this to identify clearly distinguishable coral recruits (Jonkers et al. 2008, Jonker et al. 2019).

Photographs and photo-mosaics may also be used to detect coral recruits or to make a record of the individual recruits (Hughes et al. 2007, Jonkers et al. 2008, Edwards et al. 2017). However, the two-dimensional nature of photographs may result in a greater proportion of coral recruits not being detected when relying on photographs compared to the naked eye, especially in complex habitats and communities because of the preferential settlement of corals in cryptic habitats (Hughes et al. 1999, Arnold et al. 2010). Fluorescence and ultra-violet lighting may facilitate the photographic detection of coral recruits but require more costly and delicate equipment or nocturnal surveys (Baird et al. 2006, Schmidt-Roach et al. 2008, Palmas et al. 2014). Image joining or 3D modeling techniques that can represent the structure of reef substratum are improving and may improve the ability to monitor recruit populations through time (Burns et al. 2015, 2016, Raoult et al. 2016, Edwards et al. 2017, House et al. 2018)

Coral recruitment will normally be expressed as the number of recruits per meter square ($n/m^2$). Coral recruitment can also be assessed as a rate if the abundance of coral recruits is quantified per unit of time (e.g. $n\ m^{-2}\ year^{-1}$).

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# 13. PREVALENCE OF BLEACHING

## Indicator Name

- Prevalence of bleaching

## Alternative indicator names

- Bleaching occurrence, percentage, or frequency
- Bleaching susceptible taxa
- bleaching resistant and bleaching vulnerable coral taxa (ideally species or genus level)

## Importance

High

## Definition

The prevalence of coral bleaching

## Background and Reasoning for choosing the indicator in Mozambique

Coral bleaching results from a breakdown of the relationship between a coral host and its symbiotic algae (genus: *Symbiodinium*), which results in symbiotic zooxanthellae being expelled by the coral host (Dove and Hoegh-Guldberg 2006, Oakley and Davy 2018). The symbiotic algae also contain the photosynthetic pigments that give coral a colorful appearance, without which coral tissue becomes translucent and the mostly white coral skeleton becomes visible. Coral fitness and survival are influenced by the transfer of sugars produced during photosynthesis by zooxanthellae, which provides up to 95% of the carbon requirements for growth, reproduction, and maintenance (Muscatine 1990).

It is currently recognized that global bleaching events are primarily driven by anomalous rises in seawater temperature and related to global climate changes (Heron et al. 2016b, Eakin et al. 2019). Bleaching stress is aggravated by sunlight, which stimulates disrupted photosynthetic mechanisms of zooxanthellae that result in coral bleaching (Gleason and Wellington 1993, Lesser and Farrell 2004, Rosic et al. 2020). Localized coral bleaching is also reported to have resulted from cold water stress, eutrophication, pollution, high or low salinity, and sedimentation (van Oppen and Lough 2018)

Bleaching events have progressively changed the composition of coral communities as a result of differing taxonomic susceptibility to mortality and capacity for recovery (Baird and Marshall 2002, Mcclanahan et al. 2007, McClanahan and Muthiga 2014, Hughes et al. 2018, McClanahan 2020, McClanahan et al. 2020).

Patterns in the relative susceptibility of coral taxa are reported for bleaching events (Marshall and Baird 2000, Loya et al. 2001, McClanahan 2004, Mcclanahan et al. 2007, McClanahan and Muthiga 2014). Branching colonies such as *Acropora* spp. are generally considered more susceptible to coral bleaching, whilst massive colonies such as *Porites* spp. are considered more resistant to bleaching (Marshall and Baird 2000, Loya et al. 2001, McClanahan 2004, Mcclanahan et al. 2007, McClanahan and Muthiga 2014). This hierarchy of bleaching responses amongst coral taxa is largely determined by the life history characteristics of coral taxa, and has until recently been reported to be consistent at the level of groupings of genera and family (Enríquez et al. 2017, Mizerek et al. 2018). Over longer time frames and across multiple disturbances the corals most likely to persist on a reef are those that are thermally tolerant and locally persistent colonies, remnant survivors with the capacity to regrow quickly, and regionally persistent colonies that supplied recruits (van Woesik et al. 2011).

The bleaching of a coral colony is also dependent on the stress resistance qualities of the symbiotic zooxanthellae (Baker et al. 2008, Buerger et al. 2020). This implies that progressive selection of coral-symbionts that are more resistant to bleaching can potentially result in changes in the relative susceptibility of coral taxa to bleaching (Guest et al. 2012, Baker et al. 2020, McClanahan et al. 2020). For example, higher susceptibility of *Porites* spp. than *Acropora* spp. is reported from recent bleaching events (McClanahan et al. 2020).

Overall, insight to the susceptibility of a coral community to coral bleaching can be gained from assessing the taxonomic variation in the susceptibility to bleaching within a reef community. This assessment can be based on:

- the proportion of species or genera in the community that are observed in a state of bleaching stress
- the proportion of colonies of each species or genus observed in a state of bleaching stress
- the intensity of bleaching for each species or genus calculated from the proportion of colonies in each taxa in different states of bleaching

Insights are likely to be location specific, despite the global prevalence of coral bleaching events. The frequency and intensity of coral bleaching is spatially variable in response to physical conditions that influence seawater temperature and sunlight (West and Salm 2003, Lenihan et al. 2008, 2015). Key reef features to consider include depth, shading from reef structure such as walls, upwelling seawater or nearby deep water, and tidal flushing of reef waters (Lenihan et al. 2008, 2015, Smith et al. 2014, Muir et al. 2017). Spatial variation in coral community composition may also influence coral bleaching outcomes (Graham et al. 2014, Ferrigno et al. 2016).

## This indicator is relevant to

Coral community resilience
Resilience to bleaching events

## Benchmarks and thresholds

We advise against setting a threshold for the % of bleached corals. However, reference to the frequency and intensity of bleaching for each coral taxa can be made to previous bleaching events at a reef or similar reefs within the region.

For the purpose of identifying sea water temperature thresholds that result in mortality a threshold of 50% mortality of thermally sensitive and locally abundant coral taxa was derived for six reefs on the Great Barrier Reef that suffered high mortality during bleaching events by Berkelmans (Berkelmans 2009).

## Readiness

Poor.

Few detailed coral bleaching surveys have been undertaken in Mozambique.

Available capacity (e.g. taxonomic skills) and resources may currently limit the opportunity for nationwide surveys in Mozambique during a bleaching event.

## Methods and materials

Surveys of coral bleaching may be similar to general surveys of coral abundance and diversity with the additional requirement of recording observations of the bleaching state of corals. These may be achieved simply by adding notes for the bleaching state of observed corals during line intercept transect, point intercept transect or quadrat surveys. Surveys must consider habitat features such as depth, water exchange and shading that may influence the extent of bleaching (West and Salm 2003, Lenihan et al. 2008, 2015, Muir et al. 2017).

Coral bleaching is easily noticed on a coral reef, especially during mass bleaching events, but there are several stresses and disturbances to corals that can be confused with bleaching (Marshall and Schuttenberg 2006, Obura and Grimsditch 2009). Feeding scars may be left by coral predators, such as crown-of-thorns starfish and drupella snails. These can be distinguished by patches of bare skeleton adjoining patches of live, healthy tissue. Coral diseases may result in dead or stressed coral surfaces, sometimes mistaken for the early stages of coral bleaching. A familiarity with the many visual signs of disease is recommended, but the effects of disease are often characterized by a distinct line that separates live and dead sections of a coral surface. Sedimentation may result in partial mortality of coral colonies visible as whitened patches of exposed coral skeleton when the sediment is removed. Bleaching stress usually affects entire colonies or large sections of colonies similarly, alhtough shaded areas are less likely to bleach.

It is advisable to undertake surveys that provide quantitative information for coral abundance, and if possible size frequency information, when conducting a survey to assess coral bleaching. This may be achieved using line intercept transects, belt transects or quadrats to standardize the search area in a survey (English et al. 1997, Hill and Wilkinson 2004, Marshall and Schuttenberg 2006). Photoquadrats are a useful approach to undertaking fieldwork quickly and allow for more detailed observations without the time restrictions imposed by a relatively short temporal window of bleaching stress (Edwards et al. 2017). Photographic techniques also facilitate assessing the percentage of a coral colony surface that bleaches (Chow et al. 2016). The usefulness of photographic assessments of bleaching without in situ observations may be influenced by the morphology of the corals observed. For example, massive and encrusting morphologies may be shown more clearly in two dimensional photographs, whilst these may be less useful for branching morphologies.

Bleaching events tend to be restricted to a few weeks during the warmer months of the year, thus specific rapid assessments that focus on coral bleaching will facilitate surveys of a larger number of sites (Marshall and Schuttenberg 2006, Darling and McClanahan 2016). McClanahan et al. (2001) categorically distinguished the intensity of bleaching by classifying corals as normal color, pale, partially bleached, fully bleached or recently dead. This scale was further subdivided into (1) unbleached (normal coloration), (2) pale (lighter color than usual for the time of year), (3) 0–20% of the surface bleached, (4) 20–50% bleached, (5) 50–80% bleached, and (6) 80–100% bleached (7) recently dead (McClanahan 2004). These scales have been widely used for rapid assessments in the Indian and Pacific Ocean (Mcclanahan et al. 2007, McClanahan and Muthiga 2014, McClanahan et al. 2020).

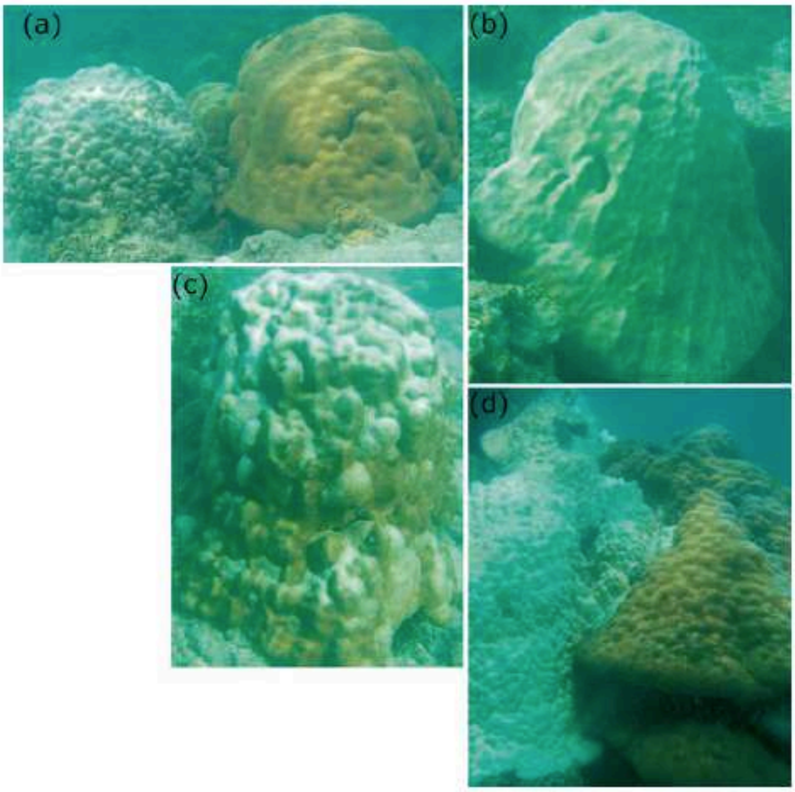

**Fig. 1a–d** Bleaching categories used in this study using examples of *Porites lutea*. **a** The coral on the left is classified as pale and is a gray or blue type of paling, whereas the one on the right is classified as normal. **b** This coral is also pale but has a yellow coloration. **c** An example of a 20–50% partially bleached coral. **d** A fully bleached coral on the left and one of normal color on the right

Attempts have been made to standardize visual assessment of bleaching intensity. Obura and Grimsditch (Obura and Grimsditch 2009) propose a method to assess coral bleaching that applies a scale originally proposed by Oliver et al. (2004), with additional reference to a colour reference card developed from zooxanthellae concentrations (Siebeck et al. 2006), recording bleaching observations as:

- 0, No bleaching evident; All shades 3-6
- B1, Partially bleached (surface/tips); or pale but not white; Shade 2
- B2, White Shade 1
- B3, Bleached + partly dead Shade 1
- D, Recently dead

The colour reference card is know as the "Coral Health Monitoring Chart" (Siebeck et al. 2006), and was developed to assess bleaching stress in corals with observation of controlled laboratory bleaching to establish relationships between colour and symbiont pigment abundance. This used five species of branching corals (*Acropora aspera, Pocillopora damicornis, Styllophora pistillata, Platygyra daedalea, Montipora digitata*), and also combined photographic field surveys of 200 corals to identify the most common colours associated with normally pigmented and bleached corals from a broader range of species.

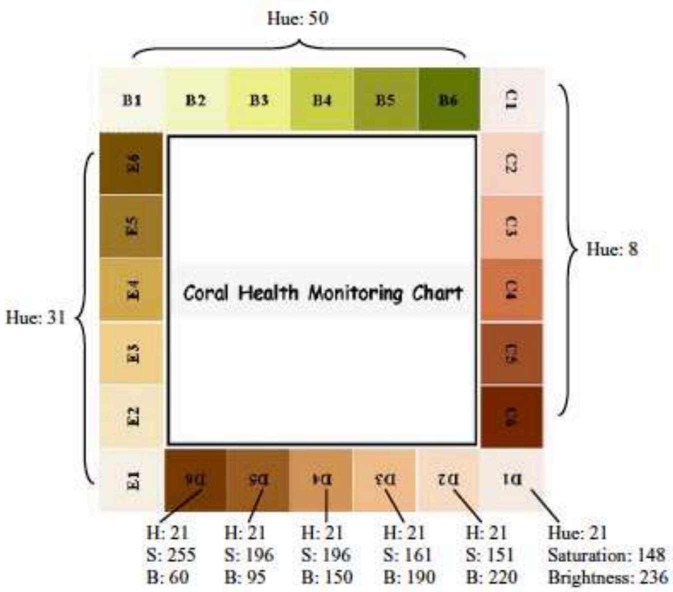

**Fig. 2** The Coral Colour Reference Card developed for standardizing changes in coral colour. The hues are given for the four *different colour* categories arranged in groups around the sides of the chart. *Brightness and saturation values* are given for one hue only since they are identical for the same numerical colour score for each of the other hues

Bleaching stress and its severity are determined not only by the magnitude of a temperature anomaly but also by the duration of the temperature anomaly (Liu et al. 2014, Heron et al. 2016a, Eakin et al. 2019, NOAA 2019). The longer corals are stressed by temperature anomalies, as well as the greater the anomaly, the more likely corals are to bleach. This means it is important to undertake surveys within as short time frame as possible during a bleaching event, because the proportion of bleached (or dead) corals is likely to increase in sites surveyed later compared to sites surveyed earlier. Records should be made of the timing of surveys and a graph or regression analysis can be used to quantify any change in the proportion of bleached corals associated with the length of a temperature anomaly.

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# 14. PARTIAL MORTALITY

## Indicator Name

Prevalence of partial mortality

## Alternative indicator names

Percent Recent Mortality (PRM)

## Importance

High

## Definition

Partial mortality refers portions of the living tissue of a coral colony that have become injured and died. This mortality may have resulted from a range of physical and biological processes, especially sedimentation and ephemeral overgrowth by algae (Hughes and Jackson 1980).

The abundance and relative frequency of coral colonies that display partial mortality provides insight to the intensity and frequency of stresses and disturbances that affect reef building corals. Therefore partial mortality also reflects the condition of a coral reef community.

## Background and Reasoning for choosing the indicator in Mozambique

Partial mortality visible on coral colonies is a record of prior stress, which may occur repeatedly (Hughes and Jackson 1980), and a higher prevalence of partial mortality in a coral community is likely to represent a poorer coral reef community condition (Nugues and Roberts 2003, Garzón-Ferreira et al. 2005, Cooper et al. 2009).

Partial mortality can provide an early indicator of stressful conditions and insight to levels of stress that can be related to a variety of stressors (Cooper et al. 2009, Lirman et al. 2014). Complete mortality does not provide early indications of coral condition or a continuum of responses to stressors, because it distinguishes only live or dead colonies (Jameson et al. 1998, 2001, Lirman et al. 2014).

Partial mortality in corals can be caused by a variety of stressors (Hughes and Jackson 1980, Nugues and Roberts 2003, Garzón-Ferreira et al. 2005, Cooper et al. 2009, Lirman et al. 2014), which include:

- sedimentation,
- physical damage from storms or human activities,
- competition (e.g. macroalgae, sponges, soft corals)
- coral bleaching,
- predation (e.g. COTS, drupella, fish), and

- disease.

Unless the agent of stress is observed to affect individual coral colonies, for example through repeated observations of tagged coral colonies, or with time interval photographs made at long-term survey sites (e.g. permanent quadrats or transects) it is only possible to speculate the exact cause of partial mortality (Hughes and Jackson 1980, 1985). For this reason historical knowledge of the disturbance and stresses to a coral reef community is important. For example, coral mortality resulting from bleaching mortality may be confused with mortality resulting from coralivorous predators (Marshall and Schuttenberg 2006, Obura and Grimsditch 2009).

It is helpful to distinguish recently dead coral from old mortality (Lirman et al. 2014). Recent mortality on a coral colony has been defined as any "non-living parts of the coral in which the corallite structures are still intact and identifiable to species" (Kramer 2003). It is usual for recent mortality to become overgrown by macroalgae within days to a week (Diaz-Pulido and McCook 2002). Percent recent mortality (PRM) provides insight to the levels of stresses to coral colonies and coral reef condition. PRM, the proportion of a coral colony that has experienced recent tissue mortality so that corallite structures in the non-living parts of the coral are still intact and identifiable to species, can be attributed to disturbances taking place within days to a few months preceding the surveys (Lirman et al. 2014).

This indicator can be easily communicated and provides (Cooper et al. 2009, Lirman et al. 2014):

- a baseline to assess the status of coral populations; and
- early-warning indicators of unfavourable conditions to trigger management actions

### This indicator is relevant to

Coral reef condition
Coral demography
Levels of disturbance or stress

### Benchmarks and thresholds

Thresholds for partial mortality have been proposed for Caribbean reefs (Lirman et al. 2014). These authors stress the importance of considering recent coral mortality and excluding old coral mortality.

PRM benchmarks (based on reef data from Florida USA from >50,000 colonies from 11 coral species and ~ 1400 sites:

- <1% PRM and <5% prevalence of partial mortality for Florida reefs during periods of background, low-stress environmental conditions.
- PRM levels >1.0% and prevalence levels >5% can be used as early warning indicators of degrading conditions.

- Average PRM values >2% are indicative of increasingly stressful conditions as those experienced during temperature anomalies and major cyclones (hurricanes).
- PRM values considerably >2% are reflective of significantly stressful conditions and warning signals of potential major coral mortality as evidenced by mean PRM levels of >10% recorded in Florida as a consequence of the 2010 extreme cold-water event.
- PRM and prevalence values from Florida reefs compared favourably with those recorded in the Caribbean and the Mesoamerican region where a benchmark of 2% for background levels of PRM under low-level, chronic stress was proposed.

## Readiness

This indicator is not readily compared between reefs because the observations required are specific and not normally included in surveys. Observers are required to possess the skill necessary for general surveys of coral abundance as well as knowledge and experience to identify partial mortality and its possible causes. It is also likely that observed levels of partial mortality are both species-specific and colony size-specific and this should be taken into consideration when making comparisons with reference coral communities or reefs.

## Methods and materials

Partial mortality can be quantified by estimating the proportion of colony surface free of living tissue, or using photographic techniques to measure the area of lesions and colony surfaces (Hughes and Jackson 1980, 1985, Nugues and Roberts 2003). This can be undertaken using dedicated software such as CPCe to quantify the areas of partial mortality on a coral surface (Kohler and Gill 2006).

Surveys of coral partial mortality in a coral reef community may be undertaken using coral colonies as the individual replicates, or searching for corals in defined areas such as quadrats or belt transects (Hughes and Jackson 1980, 1985, Nugues and Roberts 2003). The time required to make detailed observations of partial mortality in situ may complicate combining such surveys with standard surveys of coral cover or diversity unless task are partitioned between multiple observers.

Partial mortality is most easily observed on encrusting, foliacious, massive and submassive morphologies (Hughes and Jackson 1980) and observations may not be appropriate for all coral species. Tagging, mapping or photographing multiple colonies from a population of corals to undertake repeated and long-term observations is a useful approach to describe species specific trends in coral partial mortality (Connell et al. 1997, Hughes and Tanner 2000).

The following is a description of how it is suggested that partial mortality is recorded in the AGRRA protocols (Lang et al. 2010, 2013). The percentage (%) partial mortality is recorded as percent of the entire planar surface that is:

- New or recent mortality: skeletal structures are intact unless live tissues have just been bitten by a fish or abraded and have no sediment, bacterial/microalgal biofilms, turf algae, etc., on their bright white surfaces;
- Transitional mortality: skeletal structures are slightly eroded at most and covered with fine layer of sediment, microbial/microalgal biofilms, or tiny turf algae–unless they have just been bitten by a fish or abraded, thereby exposing the underlying bright white skeleton;
- Old mortality: skeletal structures are completely covered over by organisms that are not easily removed, e.g., thick algal turfs, many macroalgae and invertebrates–unless they have just been bitten by a fish or abraded, thereby exposing the underlying, bright white skeleton.

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## 15. FISH SPECIES RICHNESS

### Indicator Name

Fish species richness

### Alternative indicator names

Number of fish species in a community

### Importance

High

### Definition

Broadly speaking, species richness is the number of species in a community or region, however it may not be possible to enumerate all of the fish species in a reef community or region (Krebs 2014).

We define fish species richness as the total number of fish species in a coral reef community. For practical reasons that include differential abundance of species, differences in the detection rate of species, and different abilities of observers to detect species, the community should be specified by an area with a defined search time and sampling effort.

Species richness does not require the identification of the species counted, however the insight that can be gained into the condition of a coral reef community is increased with knowledge of the species present.

### Background and Reasoning for choosing the indicator in Mozambique

Every species may provide a unique function to an ecosystem. It is therefore reasonable to expect that increases in the species richness of a reef fish community will also increase the ecological functions of that community (Halpern and Floeter 2008). Initially it is likely that each new species encountered performs a new function, however as the number of species increases there is an increasing likelihood that species perform similar functions and there is overlap or redundancy of ecological functions. However, there are many cases of a single species or limited number of reef fish species fulfilling an ecological role (Bellwood et al. 2006, Mouillot et al. 2014).

Knowledge of species richness is frequently used in conservation and management decisions to reflect ecosystem condition. An ecosystem with more species is likely to contain more species that fulfill similar ecological functions (Walker 1992), and because species differ in their sensitivities to disturbances (Elmqvist et al. 2003), higher species richness increases the likelihood of a disturbed community retaining species to perform functional roles and maintain ecological processes. The maintenance of ecological functions arguably retains ecosystem condition more than the number of species present (Parravicini et al. 2014), although it is accepted that higher species richness is more desirable, and to some extent reflects the diversity of ecosystem functions performed by a community. Nonetheless, species richness can be indicative of the condition of an ecosystem, for example loss of reef fish species richness may reflect negative impacts of fishing (Nañola Jr. et al. 2011)

At large geo-graphical scales, variation in species richness results from the influence of factors that include, temperature, environmental stability, and geological processes (Hughes et al. 2002, Parravicini et al. 2013).
These large-scale processes establish limits to species pools and community ecologists have suggested that at local-scales the species richness and assembly of communities is a result of habitat (Gratwicke and Speight 2005, Messmer et al. 2011, Graham and Nash 2013) processes such as competition, predation, recruitment, disturbances and immigration (Wilson et al. 2006, Belmaker 2009, Nañola Jr. et al. 2011). Large scale gradients in the species richness and community structure of reef fish also result from their dispersal from center(s) of origination, or biodiversity hotspots (Elst et al. 2004, Fisher et al. 2015, Mcclanahan 2020).

There are limitations to the use of species richness to describe coral reef condition. Firstly, it is possible that communities in different condition have the same number of species, but with different combinations of species characteristic of different ecosystem conditions (Cheal et al. 2008). Secondly, to detect a change or difference in species richness using data from counts of species, assumes that all species are detected, and that the detectability of the different species is the same or at least does not differ between the surveys or communities that are compared. Detectability of a species is the probability of detecting at least one individual of the species in a survey, provided at least one individual of that species is present in the area of interest during the time of the survey (Boulinier et al. 1998). Field surveys rarely detect all of the species that exist in a community, especially when these are motile such as reef fish (Krebs 2014, Fisher et al. 2015).

Overall species richness is most useful for describing the condition of fish community on a coral reef when combined with other information, such as species heterogeneity and evenness (Krebs 2014). This is further complemented by knowledge of the species present, the relative abundance of species present, and the size frequency distributions of fish which facilitates estimates of fish biomass (McClanahan 2018, 2019a, Froese and Pauly 2020).

### This indicator is relevant to

Coral reef condition
Community diversity
Coral reef fisheries
Coral reef resilience

## Benchmarks and thresholds

The total number of reef fish species in the Western Indian Ocean is over 900 species and is relatively high when compared to other regions (Biodiversity and Biology 2003).

It is difficult to estimate an average fish species richness for coral reef sites in Mozambique because most surveys in this region have not identified fish to the level of species. A total of 263 species were reported to be present in a single survey of reefs shallower than ~15 m at Vamizi Island in the North Quirimbas, from where 373 species of reef fish have been reported (Hill et al. 2009)

Species richness of coral reef fish has however been associated with reef fish biomass on reefs of Eastern Africa, including Mozambique (McClanahan 2019b). Fish species richness is estimated to average 50 species on coral reefs with a fish biomass of ~ 1000 kg per 500 m$^2$ and 45 species on reefs with a fish biomass of ~ 500 kg per 500 m$^2$.

Therefore we suggest that a species richness of coral reef fish, on a reef with fish biomass of at least 1000 kg per 500 m$^2$ and surveyed to a maximum depth of 20 m for a site of 500 m$^2$ area, may represent a coral reef condition that is:

- Good, when species richness is 50 or greater
- Average, when species richness is between 45 and 50 species
- Poor when species richness is below 45 species

Note that a coral reef with reef fish biomass lower than 500 kg per 500 m$^2$ is likely to be in poor condition (McClanahan 2019b), thus we avoid suggesting species richness thresholds for this situation.

However, we believe these thresholds are likely to be adapted in response to improved monitoring of coral reefs in Mozambique, and we recommend caution and careful consideration of natural habitat features that may result in lower or higher species richness when applying these suggested thresholds.

## Readiness

Medium to low.

The indicator does not require fish species to be identified, although this is benefitial. This requires skilled and experienced observers and is not widely undertaken in Mozambique.

It is also necessary for different studies to use the same methods for estimates of species richness to be comparable.

## Methods and materials

Species richness is the number of different species in a community and is estimated from a counted in a sample of a community. The sampling effort is described quantitatively with reference to the time (e.g. minutes in a timed swim), the number of replicates used to sample the community (e.g. video point samples, UVC transects), the distinct habitat or the size of the area surveyed.

Species richness can generally be estimated from any method used to survey reef fish that identifies the species present at a location, such as from underwater visual census (UVC) using either belt transects or point surveys, or underwater video surveys (English et al. 1997, Samoilys and Carlos 2000, Labrosse et al. 2002, Hill and Wilkinson 2004, Mike Cappo et al. 2014). It is important to define both the search effort and area

Initially increasing the number of replicate observations or individuals sampled increases the likelihood of including a new species. As the number of species identified increases the probability of identifying a previously unrecorded species decreases. The shape of this relationship between the search effort (e.g. number of transects, number of video samples, ha of reef) and increases in the number of recorded species can be used to describe the species richness of a community and compare communities surveyed with similar methods (Krebs 2014). Using similar methods is important because species richness estimates are affected by the method used to sample a community (e.g. drop camera, diver surveys etc), which all have their inherent biases (Samoilys and Carlos 2000, Mike Cappo et al. 2014, Wilson et al. 2018).

An important consideration is that ideally the sampling effort upon which species richness is estimated should be standardized (Gardener 2014). The larger the area that is sampled, or the longer species are searched for in a given area, the more species are likely to be identified. When sampling effort differs, methods to standardize species richness such as rarefaction, jackknife estimates or bootstrap procedures that take into account different sampling effort should be applied (Gardener 2014, Krebs 2014)

Species richness can be plotted as a species-accumulation curve, which displays the number of samples (e.g. transects) along the x-axis and the total number of fish species along the y-axis (Krebs 2014). Species-accumulation curves can be used to compare patterns of diversity between communities. The species-accumulation curve also provides an indication of how many samples are required to describe species richness when the curve becomes asymptotic. It is also important to realize that it is unlikely that all of the species present in an area will be identified (Krebs 2014) even if the number of new species identified declines with each new sample.

However modern video techniques used to sample a reef fish community provide a permanent record with improved resolution which may allow for repeated observations or consultations with colleagues and identification resources provide a more complete count of the species present (Colton and Swearer 2010, Wilson et al. 2018). Videos may also be deployed os as to survey reef fish without the presence of a person to have less influence on fish behavior and detectability (Dickens et al. 2011).

Statistical software may also be used to predict species richness curves based upon random samples (Krebs 2014). A variety of statistical software packages and packages in the R language have the capacity to estimate species richness (e.g. vegan, SPECIES, rich, SpadeR, BiodiversityR). Rarefaction can be used to adjust for variation in species abundance and sampling effort, and for example in the package vegan this is undertaken with the command rrarefy().

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# 16. FISH ABUNDANCE, SIZE AND ESTIMATED BIOMASS

## Indicator Name

Fish abundance, size and estimated biomass.

## Alternative indicator names

None to our knowledge.

## Importance

Critical

## Definition

The number and estimated size of individual fish, identified to taxonomic levels of species, genera or family, and observed in a defined area of coral reef habitat (e.g. 1 hectare; 500 $m^2$). Biomass is estimated from calculations based on predetermined length-weight relationships for each species. Species can also be grouped into functional groups.

## Background and Reasoning for choosing the indicator in Mozambique

The abundance, community composition and biomass of coral reef fish communities are generally evaluated together to assess the condition of a reef fish community (Wilson et al. 2010, Graham et al. 2017a, McClanahan and Muthiga 2017). The abundance and size of fish are observed and recorded simultaneously in field surveys of a defined area, usually a belt transect (English et al. 1997, Samoilys and Carlos 2000, Labrosse et al. 2002a, Hill and Wilkinson 2004). This information is used to estimate the biomass of fish in an area (Froese and Pauly 2020). Non-cryptic fish are often an accepted representation of the reef fish community, particularly when there is a fishery focus, and surveys may focus on fish grouped by families or genera that represent most of the fish biomass in a reef fish community (McClanahan and Muthiga 2017).

Fish biomass and community structure reflect human impacts such as fishing pressure and habitat degradation (McClanahan 2019, McClanahan et al. 2019a). Field studies have generally found that coral reef fish density and biomass are influenced by coral reef condition, community composition and environmental factors such as live coral cover (Coker et al. 2014, Pratchett et al. 2018), coral reef complexity or three-dimensional structure (Wilson et al. 2007, Graham and Nash 2013, Richardson et al. 2017), reef habitat type and depth (Acosta et al. 2014, Williams et al. 2015, McClanahan and Muthiga 2017, Samoilys et al. 2018).

Identification of fish also allows their separation into functional and trophic groups which can enable the interpretation of ecological interactions with ecosystem level effects, such as predation or herbivory (Green et al. 2009, Froese and Pauly 2020). For reef fishes, trophic groups are generally synonymous with groups of species from different trophic levels within a food chain (e.g. predators and herbivores), reflecting the nature of their diet, feeding behavior and their influence on energy flow in reef ecosystems. Functional groups gather species that perform a similar ecological function (e.g. grazing) that influences ecosystem processes, irrespective of their taxonomic affinities (Bellwood et al. 2004, 2019). Therefore reef fish surveys usually assess taxonomic diversity, size structure and biomass together and later subset datasets to gain specific insight to ecological condition and process of a reef.

### This indicator is relevant to

Coral reef condition,
Coral reef resilience,
Ecosystem function,
Coral reef fisheries

### Benchmarks and thresholds

The suggestions of benchmarks or thresholds for the abundance of coral reef fish is usually undertaken with reference to fish biomass within an area (Darling and D'Agata 2017, Graham et al. 2017b, McClanahan  Timothy R 2018, McClanahan et al. 2019b, 2021)

Benchmarks for fish biomass at reef sites of the East African coast, including for Mozambique and in habitats shallower than 20 m, are proposed as (McClanahan 2018, 2019, McClanahan et al. 2019a):

- 1150 kg/ha for desirable conservation outcomes,
- 600 kg/ha to maintain diversity in fish communities, and
- 450 kg/ha to allow for artisanal fishing to be sustainable.

Recent examples of coral reefs shallower than 20 m in the Western Indian Ocean that are likely to be in very good condition suggest average reef fish biomasses of (McClanahan et al. 2021):

- 2,450 kg/ha (95% confidence interval, 95% CI, 2,130–2,770 kg/ha; median = 1,885 kg/ha), on reefs in remote areas without people (>9 hours of travel time from urban areas).
- 910 kg/ha (95% CI 823–989 kg/ha), on reefs in high-compliance closures or marine protected areas, that have been protected for 20 years and for which the protected area is 5–10 km$^2$.

### Readiness

Medium

Fish surveys to gather biomass data require relatively experienced and skilled observers, however biomass of reef fish is a widely collected metric that is readily compared between reefs and regions. Surveys of coral reef fish frequently aim to obtain this information.

## Methods and materials

Underwater visual census (UVC) of reef fish using visual belt transect surveys are a widely accepted and standardized method for assessing the abundance and diversity of fish communities (English et al. 1997, Samoilys and Carlos 2000, Labrosse et al. 2002a, Hill and Wilkinson 2004, Dickens et al. 2011). In this method a snorkeler or SCUBA diver team lays a transect, or replicate transects, usually of 50 m length and makes observations along these within a defined width of 5 m either side, from this example to survey 500 m$^2$ (English et al. 1997, Samoilys and Carlos 2000, Labrosse et al. 2002a, Hill and Wilkinson 2004). Water visibility is ideally of 10 m of more to facilitate seeing, counting and accurately identifying fish. Records are made of the observed fish that include their estimated size, and identifying these to the level of family, genus or species according to the skill and the focus of the survey. A survey may focus on key species targeted by fisheries (McClanahan et al. 1999, Cinner et al. 2016, Graham et al. 2017b). Estimates of the size of the fish and the number of fish observed for each size are required to estimate biomass (Labrosse et al. 2002b, Froese and Pauly 2020). Note that vision is distorted underwater and objects appear closer and larger, therefore observers should train, or routinely calibrate, their ability to estimate fish size with fish silhouettes of known sizes that represent the variety of fish morphologies (English et al 1997). There are also biases associated with underwater visual censuses of reef fish (UVC) that relate to human presence and how this influences fish behavior (Dickens et al. 2011, Wilson et al. 2018) and the detectability of reef fish (Bozec et al. 2011). Together this information allows for the estimation of fish biomass in an area of habitat, for example in units of kg/ha of fish.

If the focus of observations are rare taxa such as sharks it is advisable to increase the area sampled by extending the length and width of transects (e.g. 500 m length and 20 m width). Baited video techniques are also frequently used, however these do not ample a defined area so only provide relative abundance estimates and also bias sampling to include greater abundance of predators (Colton and Swearer 2010, Mike Cappo et al. 2014, Goetze et al. 2019)

Belt transect surveys of reef fish require a relatively skilled SCUBA diver who is also skilled and experienced at fish identification. The observer should progress along a transect at a steady speed, paying attention not to count the same individual fish more than once. It is normal for the primary observer’s diving buddy to lay the transect tape as the fish are surveyed, potentially making secondary observations, but to lay a transect tape first and then return to survey fish would result in bias by scaring timid fish from the site (Labrosse et al. 2002a, Hill and Wilkinson 2004, Dickens et al. 2011)

The primary observer identifies and estimates the size of all fish inside the area of the belt transect. Fish smaller than 3 cm are ignored, unless there is a specific interest in juveniles, which requires greater skill and experience. Alternatively for practical reasons, and to focus on fish more relevant to fisheries, observers may ignore fish smaller than 10 cm, but this should be decided after an appraisal of the size of fish caught in local fisheries.

Two divers are necessary for safety, and it is not advisable to pair a diver observing fish with a diver observing benthic organisms. Usually observers swim one direction counting the larger fish which are more motile and likely to be scared from the survey area by a divers presence (Dickens et al. 2011), then observers count the smaller territorial fish (e.g. damsel fish) in reverse direction along the transect (English et al. 1997, Samoilys and Carlos 2000, Labrosse et al. 2002a, Hill and Wilkinson 2004). One diver can count larger fish and the other smaller fish (e.g damsel fish) or simply lay the transect tape.

The multiple observations made during a fish belt survey can make written records confusing and it is advisable to use a well planned and previously printed data sheet (English et al. 1997, Hill and Wilkinson 2004).

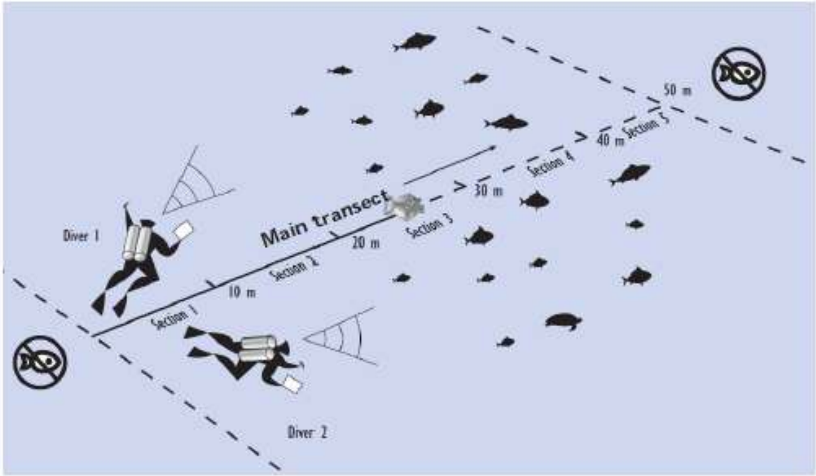

**Figure 1. Image of a pair of divers undertaking a fish belt transect survey** (Labrosse et al. 2002a).

After fieldwork it is necessary to organize the data collected and perform calculations to estimate fish biomass. In its simplest form the biomass of individual fish is calculated using the allometric length-weight conversion:

$$W = a \times TL \times b$$

**Equation 1. Fish allometric length-weight conversion**

Parameters a and b are species-specific constants, TL is the individual total fork length in cm of a fish and W is the fish weight in grams (Froese and Pauly 2020). Fish biomass values are converted to kg/ha to enable standardized comparisons with other regional studies (Graham et al. 2017a).

Dedicated data entry platforms such as Mermaid (www.datamermaid.org) aid in the post fieldwork processing of reef fish data by automatically correcting species name entries, highlighting unusual fish sizes to be checked, and calculating the biomass values for each observed fish using the widely accepted a and b parameters listed in FishBase (Froese and Pauly 2020).

The size and abundance of reef fish has traditionally been surveyed visually by a skilled and experienced observer, which enables global comparisons (Cinner et al. 2016, Graham et al. 2017a), and is favoured by the low cost of the equipment required (e.g. 50 m survey tape, slate and data sheet, SCUBA equipment) despite requiring skilled personnel (English et al. 1997, Labrosse et al. 2002a, Hill and Wilkinson 2004, Wilson et al. 2018). Video methods may be more costly and require time consuming post fieldwork examination, however can extend the spatial (e.g. deeper than SCUBA limits) and temporal possibilities of fish surveys (Mike Cappo et al. 2014). The development of photogrammetry and algorithms to identify fish species from images is also increasing the opportunities to standardize and increase the replication of surveys of reef fish with video (Shortis et al. 2016, Villon et al. 2016, 2018). However, these methods currently require corrections for the differences in detection of specific size groups and taxonomic groups of fish between the human eye and video and human observation versus automated observation (Wilson et al. 2018). Diver operated video systems differ to single point video systems because they offer the ability to survey a defined area (e.g. belt transect). However it is important to standardize the procedures for their use (Goetze et al. 2019).

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# 17. FISH COMMUNITY FUNCTIONAL STRUCTURE

## Indicator Name

Composition of fish population or community structure by functional/trophic groups.

## Important subdivision of this indicators

Special attention is often given to:

- Fish that feed on coral of fish that depend on live coral habitat
- Fish that feed on macroalgae
- Top order predators, that feed on other fish

## Alternative indicator names

Reef fish functional diversity
Reef fish trophic structure
Trait diversity of reef fish

## Importance

Critical

## Definitions

Functional group: A collection of functionally similar species that have a similar functional role in a system regardless of morphological or behavioural characteristics (traits), or taxonomic affinity (Bellwood *et al.* 2004, modified after Steneck & Watling 1982; Steneck 1983). A key element of understanding functional groups is to identify "what" impacts a species has on the ecosystem and community, for example removing upright macroalgae, removing algal turf, eroding carbonate structure or removing sediments (Bellwood et al. 2019).

Trait: An attribute, feature, characteristic of an organism which can be morphological, physiological or behavioural (Viole et al 2007).

Functional trait: A trait that has a demonstrable, causal, relationship with a specified function (Bellwood et al. 2019).

Ecosystem function: The movement or storage of energy or material in an ecosystem (Bellwood et al. 2019).

## Background and Reasoning for choosing the indicator in Mozambique

The functional structure of an assemblage, or the ecosystem functions that are performed by the reef fish species present in an ecosystem, is a key indicator of the condition or resilience of an ecosystem (Bellwood et al. 2019, Matthew et al.

2019). This is recognized in conservation tools such as the IUCN Red List, which highlight that loss of endangered species may dramatically change the functioning of ecosystems through the alteration of functional diversity (Gaston and Fuller 2009). On a coral reef each species of fish possesses an individual combination of traits that play a role in determining their performance, resource use and ecological impacts (Bellwood et al 2004). As a result each species has the potential to influence other organisms in the coral reef community and ecosystem as a whole. The traits of each species provide insight into "how" this influence may occur (Wainwright and Bellwood 2002, Mouillot et al. 2013, Bellwood et al. 2019, Matthew et al. 2019). However, understanding "what" impacts a species actually has on the community or ecosystem is a more complex outcome influenced by factors that include behavior and environment (e.g. temperature), a distinction that has somewhat confused the study of functional ecology (Bellwood et al. 2019).

A well developed and frequently used example of grouping coral reef fish by their function has been to distinguish herbivores, or fish that feed on and remove macroalgae (Green et al. 2009). Herbivores play an important role in maintaining coral reef condition and resilience (McClanahan et al. 2012, Ceccarelli et al. 2018), through altering habitats and indirect effects on benthic the coral reef community, such as influencing interactions between macroalgae and corals (McCook et al. 2001, Birrell et al. 2008), or the dynamics of predator prey interactions (Catano et al. 2016) or even fish recruitment (Wilson et al. 2010, Evans et al. 2014).

Despite the acknowledged importance of functional groups of coral reef fish studies have not been consistent with how fish are separated into functional groups (Halpern and Floeter 2008, Bellwood et al. 2019). Classifications largely based on dietary traits have resulted in 14 functional groups of reef fish that are widely used (Table 1) (Bellwood et al. 2004, Ferreira et al. 2004). FishBase contains dietary descriptions for 2,319 species of coral reef associated fish (Froese and Pauly 2020). These dietary descriptions are based on numerous localized studies of fish stomach compositions or observation of fish behavior (Kulbicki et al. 2005, Cole et al. 2008), and it is assumed the findings are applicable to other locations. Alternatives have been to classify reef fish into functional groups based on proxies (Halpern and Floeter 2008). One proxy is maximum body size which influences predator-prey interactions, feeding behavior, the size and type of food that can be consumed (Kulbicki et al. 2005). Another proxy is maximum depth which determines spatial access to food or prey. However all of these approaches have limitations in that they intuitively infer the function undertaken by a species do not directly observe the functions species perform (Bellwood et al. 2019), and the functions reef fish perform are not always intuitively associated with their traits (Bellwood et al. 2006a).

**Table 1. Fourteen frequently used functional groups for coral reef fish that are based on dietary characteristics** (Bellwood et al. 2004, Ferreira et al. 2004, Halpern and Floeter 2008)

| Functional Group | Description |
|---|---|
| Macrocarnivores | Fish that eat mobile benthic organisms and fish |
| Strict piscivores | Fish that eat only fishes |

| | |
|---|---|
| Mobile benthic invertivores/cleaners | Fish that eat primarily crabs, molluscs and other benthic mobile invertebrates |
| Sand invertivores | Fish that eat primarily benthic invertebrates in the adjacent sandy areas to the reefs |
| Coral/colonial | Fish that eat sessile benthic invertebrates sessile invertivores |
| Spongivore/herb | Fish that eat primarily sponges, but also some algae |
| Diurnal planktivores | Fish that eat primarily macro- and micro-zooplankton during the day |
| Nocturnal planktivores | Fish that eat primarily macro- and micro-zooplankton during the night |
| Territorial algae/detritus | Fish that eat farmed turf-algae within their territories |
| Turf grazing | Roving herbivorous fish that feed removing only algae and associated epiphytic material |
| Scrapers | Roving herbivore that removes pieces of the substratum together with algae, leaving distinctive but relatively shallow bite scars |
| Excavator/eroders | Roving herbivore similar to scrapers but leave deeper bite scars and remove greater quantities of substrata with each bite |
| Macroalgae browser | Roving herbivore that tends to bite or 'crop' the algae leaving the basal portions intact |
| General omnivores | Fishes that eat a variety of organisms, including both animal and plant material in similar quantities |

Grouping coral reef fish has traditionally been achieved by referring to morphological or phylogenetically characteristics, but the functional impact a coral reef fish may have on the ecosystem is also a result of less easily defined, and spatially variable traits such as behavior (Bellwood et al. 2019). For example, herbivores may be subdivided into functional groups based upon "how" they feed on macroalgae as well as "what" impact they have on the community or ecosystem (Table 2). Such detailed understanding of the functional role of coral reef fishes is limited, and is most well understood for herbivores and corallivores which have received substantially more research attention (Pratchett et al. 2018, Bellwood et al. 2019). Understanding biological interactions such as herbivory or coralivory from a functional point of view may be improved by also considering the influence of accepted functional groups of macroalgae (Steneck and Dethier 1994) and corals (Bellwood et al. 2004).

**Table 2. A detailed functional group matrix for coral reef fishes with an average total length >20 cm, that are roving herbivores present in the Indo-Pacific** (Bellwood et al. 2019).

| 'What' functional grouping or ecosystem function | 'How' functional groupings | Description | Associated traits | Example species |
|---|---|---|---|---|
| Brushers – | Brushers | Brush | Bristle-like teeth, | *Ctenochaetus* |

| | | | | |
|---|---|---|---|---|
| remove particulates from the epilithic algal matrix but not algae | | particulate material from algal turfs. | gizzard-like stomach | *striatus* |
| Grazers – remove algal material (<2 cm) from coral reefs | Croppers (Concealed) | Can nip-off short algae from concealed microhabitats. | Multi-denticulate teeth, elongated snouts, pairing behaviour | *Zebrasoma scopas, Siganus vulpinus* |
| | Croppers (sand 'suckers') | Remove algal material and particulates. | Multi-denticulate teeth, gizzard-like stomach | *Acanthurus olivaceus* |
| | Croppers (Open) | Nip-off short algae in exposed microhabitats. | Multi-denticulate teeth, thin-walled acidic stomachs | *Acanthurus lineatus* |
| | Scrapers | Scrape the outside of the reef matrix. | Beak like jaws with gracile associated musculature, pharyngeal mill | *Scarus rivulatus* |
| | Excavators | Bite into the reef matrix. | Beak like jaws with robust associated musculature, pharyngeal mill | *Bolbometopon muricatum* |
| Browsers – remove algal material (>2 cm) from coral reefs | Chopper | Remove the entire algal thallus. | Rod/peg-like teeth, robust adductor mandibulae muscle | *Kyphosus vaigiensis, Naso unicornis* |
| | Shearer | Only nibble on leafs. | Bicuspid teeth, small gape, light adductor mandibulae | *Siganus canaliculatus* |

The interpretation of the functional structure and ecological impacts of a coral reef fish community can be undertaken by inspecting the distribution of fish biomass or abundance across functional groups. Functional structure has been used to provide insight to the overall ecological condition and functioning of a coral reef communities (Graham et al. 2011), habitat distribution of fish (Richardson et al. 2017), global biodiversity of reef fish (Bellwood et al. 2002), the impacts of human activity such as fishing and restrictive management (McClanahan and Hicks 2011, McClanahan 2019, Mbaru et al. 2020), and the resilience of a coral reef community (Green et al. 2009, Fong et al. 2018).

When the ecological functions undertaken by coral reef fish are intact a coral reef ecosystem is likely to be in better condition and more resilient. However, it is important to acknowledge that a reef fish community does not always have representatives from all possible functional groups. The patterns of presence and absence of functional groups may result from either evolutionary or ecological processes, such as historical extinctions, species radiations, barriers to dispersal

(Bellwood & Wainwright 2002) and ecological processes that govern the success and distribution of species (Halpern and Floeter 2008).

Ecological functions are most stable when each function is assured by multiple species. The overlap of ecological functions by multiple species is referred to as functional redundancy, and the greater the ecological redundancy the more likely that at least one species will resist disturbances to perform an ecological function. A challenge associated with evaluating the functional redundancy of a coral reef fish community is to precisely define relevant ecological functions (Bellwood et al. 2019). For example Table 2 exemplifies the importance of subdividing herbivorous fish and it is clear that the term herbivore refers to multiple ecosystem functions and feeding habits (Green et al. 2009). The most accurate approach to identifying the ecological functions and the species that perform functional roles is to observe these in the coral reef ecosystem of interest.

Species richness is associated with ecosystem functions but this should be undertaken with caution and attention to local details. In communities with low species richness, the likelihood that each species performs a distinct ecological function is higher, but as species richness increases there is a greater likelihood of overlap in ecological functions and greater functional redundancy (Halpern and Floeter 2008). Nonetheless, high functional redundancy among species is not necessarily related to high species richness (Mouillot et al. 2013, Brandl and Bellwood 2014). Rare species often possess distinct combinations of traits and species that have low functional redundancy often support the most vulnerable functions, with overlap of traits or functions in other species (Mouillot et al. 2013). It is also suggested that a small subset of the herbivore assemblage account for the majority of browsing on coral reefs in the Atlantic, Indian and Pacific Ocean (Topor et al. 2019). Therefore high species richness does not necessarily guarantee ecosystem functioning and loss of a single or few species might result in loss of ecosystem functions.

### This indicator is relevant to

Coral reef community condition
Coral reef community resilience
Sustainable fisheries
Ecosystem function

### Benchmarks and thresholds

For practical reasons it is generally assumed that the functional structure of a coral reef fish community is associated with reef fish biomass, and generally for surveys of habitats shallower than 20 m depth (Graham et al. 2017).

Benchmarks for fish biomass at reef sites of the East African coast, including Mozambique, are proposed by McClanahan (2019) as:

- 1150 kg/ha for desirable conservation outcomes, that translate to maintenance of the greatest number of functional groups in a reef community
- 600 kg/ha to maintain diversity in fish communities, although likely loose a large proportion of higher order predators (Graham 2017)
- 450 kg/ha to allow for artisanal fishing to be sustainable.

Therefore, we suggest that a coral reef in good condition, with an intact functional structure of reef fish requires a biomass of 1150 kg/ha; below this the functional structure of reef fish is likely to become increasingly impoverished and the reef condition poor.

Benchmarks for the number of functional groups in a coral reef will require local study to identify the local importance of ecological functions and the species that perform them (Bellwood et al. 2019).

### Readiness

Assessment of the functional composition of a reef fish community is both feasible and readily comparable to reefs in other regions.

The survey skills required to identify reef fish and conduct accurate surveys may be the most limiting factor.

### Methods and materials

The methods and materials for the assessment of reef fish functional groups are initially similar to assessments of reef fish abundance, size and estimated biomass. Ideally field surveys of reef fish identify fish to species level, although frequently to genus or family level (McClanahan et al. 2008, McClanahan and Muthiga 2017). Accurate identification of reef fish makes it possible to later separate and interpret data by functional or trophic groups based on prior knowledge of the life history traits and life strategies of each taxa.

The understanding of ecological functions and classification of reef fish into functional groups is a developing field. On the one hand, for consistency it is advisable to use widely accepted functional groups for reef fish, (Bellwood et al. 2004, Ferreira et al. 2004, Halpern and Floeter 2008), or widely accepted dietary habits listed in FishBase (Froese and Pauly 2020). We recommend the use of data mermaid (www.datamermaid.org) to enter data from surveys of reef fish, because this will automate the allocation of fish taxa to dietary and functional groups based on traits listed in FishBase. On the other hand it is important to recognize that it may be necessary to adapt widely accepted classifications of ecological function and functional groups, to best represent local circumstances. It is critical to identify the ecosystem functions relevant to a coral reef, and to recognize that the species of coral reef fish that contribute to each individual ecological function may be taxonomically distinct and possess different traits (Bellwood et al. 2019). This implies that local ground truthing of ecological functions and the species that contribute to them is necessary, and the following four steps are recommended in decreasing order of importance (Bellwood et al. 2019):

1) identify the critical ecosystem function(s), (e.g. removal of sediment) at the appropriate spatial and temporal scale for the system of interest,
2) at a matching scale, identify the traits that are important for mediating this function (e.g. scraping teeth to remove sediments from turfs),
3) identify the species that possess the requisite traits and deliver the required functions (e.g. parrot fishes, surgeon fishes, etc).
4) identify and quantify critical functional groups, which are collections of species with appropriate traits that deliver the required ecosystem function

The observation of ecological functions of a coral reef can be achieved with use of remotely deployed video cameras to observe reef fish without influences to their behavior that result from human presence (Bellwood et al. 2006b). This should complement underwater visual census (UVC) of reef fish, which remains the most practical survey method because it is widely used and cost effective for experienced observers (Dickens et al. 2011). Modern applications of photogrammetry have made it possible to obtain accurate estimations of the size frequency structure of fish populations (Willis et al. 2000, Watson et al. 2005, Costa et al. 2006, Wilson et al. 2018). However, stereo video or photography imagery is usually gathered from a single location and is therefore not suitable for estimating biomass in an area of reef. Alternatively a person can swim a stereo video camera system or these can be mounted to an automated underwater vehicle (Wilson et al. 2018). Each method has associated biases, which can limit comparison of results between them (Willis et al. 2000, Wilson et al. 2018).

Automated identification of reef fish from videos is a developing field, although currently cannot provide resolution to species (Shortis et al. 2016, Saleh et al. 2020). Therefore automated identifications is currently not ideal for enabling the accurate distinction and monitoring of functional groups of reef fish.

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# 18. FISH SIZE-FREQUENCY DISTRIBUTION

## Indicator Name

Fish size-frequency distribution

## Alternative indicator names

Fish size classes, fish size class distribution, fish population size distribution, fish population size structure, fish size-class frequency

## Importance

High

## Definition

The size-frequency distribution of a reef fish population is the description of the frequency (or number) of fish distinguished in size classes. The frequency of fish in different size classes and the change in this distribution over time gives insight into the demography and dynamics of a reef fish population (e.g. rates of recruitment, growth reproduction and mortality). This information is useful in assessing the condition of a coral reef fish population, or community when the populations of several species are assessed.

## Background and reasoning for choosing the indicator in Mozambique

Knowledge of the size frequency distribution of fish populations provides various insights into the condition of a coral reef fish community. The size frequency of coral reef fish provides insight to the impacts of fishing (Jennings et al. 1999, Zgliczynski and Sandin 2017), specific impacts of fishing gears (Mbaru et al. 2020), interspecific interactions (Kulbicki et al. 2005) and ecosystem functions (Halpern and Floeter 2008).

Records of the size of reef fish observed are necessary to estimate the biomass of a reef fish community from survey data (Labrosse et al. 2002, Kulbicki et al. 2005, Froese and Pauly 2020). Whilst, the biomass of a reef fish is one of the most frequently used metrics to assess the benefits of management regimes, impacts of fisheries and to compare reefs regionally (Graham et al. 2017, McClanahan 2019, McClanahan et al. 2019). Larger fish are targeted preferentially and the size of fish present in a community frequently decreases as fishing pressure increases (Jennings et al. 1999, Jennings 2005, Wilson et al. 2010, Graham et al. 2017).

Size structure gives insight into the reproductive potential of a reef fish population. Fish must reach a threshold minimum size before they become reproductively active (Chambers and Trippel 1997, Pears et al. 2006, Trippel 2013). Size also determines the relative reproductive output of fish, with larger and more experienced fish producing proportionally more larval offspring than smaller reproductive fish (Chambers and Trippel 1997, Berkeley et al. 2004, Pears et al. 2006, Hixon et al. 2014).

Certain species of reef fish change their diet as they grow, which enables size classes to be used to distinguish the ecological functions fish perform on coral reefs. Examples include *Naso elegans, N. tuberosus, N. unicornis,* and other *Naso spp*., which when <21 cm (<16 cm for *N. hecacanthus*) are herbivores, that browse on large macoralgae, but change their to feeding on plankton in the reef water column as the become larger (Samoilys et al. 2019 and references therein). Herbivore parrotfish in the genus *Chlorurus* also change their feeding behaviour at 35 cm, from small excavators that remove algae and substrate but play a smaller role in bioerosion, to large excavators that take few, large, deep bites, and remove calcareous substratum playing a large role in bioerosion (Samoilys et al. 2019 and references therein).

Fish body size and resulting morphological traits fish also influence biological interactions, such as interactions between predators and prey by determining morphological traits that affect feeding behavior, swimming ability, relative sizes and the types of food that can be ingested (Jennings et al. 2002, Kulbicki et al. 2005, Halpern and Floeter 2008). Based upon the maximum size reported for fish species in the Atlantic and how size might influence biological interactions 4 size categories (<10, 10 – 25, 25 – 50, and > 50 cm) were proposed as a proxy for ecological function when combined with other trophic groups or depth distribution (Halpern and Floeter 2008). These simple combinations of size, trophic group and depth distribution enabled a detailed appraisal of the condition of fish communities, providing a strong example of how knowledge of fish size-class distribution can be useful.

### This indicator is relevant to

Coral reef resilience
Coral reef condition
Coral reef fisheries
Fish reproduction and predation

### Benchmarks and thresholds

To identify specific benchmarks of size-class distribution for coral reef fish populations is complicated by the need to address each species individually, and by limited knowledge of species-specific life history traits. This often results in generalizations to the most closely related species (e.g. by genera or family). For example, Halpern and Floeter (2008) suggest four general size classes as a proxy for interpreting fish interactions amongst fish in the Atlantic Ocean (<10 cm, 10 – 25 cm, 25 – 50 cm, and > 50 cm).

Specific knowledge for individual species identifies minimum known sizes of reproductive individuals and the sizes that have maximum reproductive output (Pears et al 2006), or sizes when feeding behavior changes (Samoilys et al. 2019). Therefore it is important to consult the scientific literature and databases such as FishBase (Froese & Pauly 2020) to identify when this knowledge is available for species of specific interest.

It is beyond the current scope of this document to tabulate size thresholds for each of the species of reef fish known for Mozambique. Therefore, we recommend a stepwise procedure of:

1) Identify the fish species present at a reef site.
2) Distinguish the species present that are likely to perform important ecological functions, and which influence the condition of a coral reef based on the published literature (e.g. Green et al 2009, Samoilys et al 2019, Bellwood et al 2019). An example of the ecological functions undertaken by reef fish from the Western Indian Ocean, including Mozambique was prepared by Samoilys et al. (2019) from which Table 1 has been adapted.
3) Consult the scientific literature and FishBase (Froese and Pauly 2020) to identify species specific size thresholds relevant to behavior, diet, and reproduction.

The shape of a size class (x-axis) by frequency (y-axis) distribution graph may also provide insight to the condition of a coral reef. The size-frequency distributions of a fish population in good state, with unhindered reproduction and recruitment and that has not suffered acute mortality as a result of disturbance events is likely to be positively skewed, having a high number of small fish and progressively less fish as size increases. A negatively skewed fish population (i.e. with few recruits and increasing more larger fish) is likely to be recruitment limited suggesting there is a deleterious influence on fecundity of a population or the supply and survival of recruits to a reef. Historical knowledge of local population size-frequency distributions for key species (e.g. *Serranid spp*., *Lujanus spp*., *Chlorurus spp*., *Naso spp.*, *Acantharus spp.*, *Chaetodon spp*.) can help to assess the condition of a reef fish community and identify or predict the likelihood of a coral reef community persisting at least in its current state under local environmental conditions and stresses. Changes in the size-frequency distribution of a reef fish populations over time will also provide insight into the dynamics and condition of a coral reef community.

**Table 1. Trophic groups and functional characteristics of key reef fish from reefs in the Southern Western Indian Ocean** (Samoilys et al. 2019)

| Functional group | Description of feeding habits and reasons for selection | Group/family | English name or species |
|---|---|---|---|
| **Piscivores** | Top-level predators, exert top-down control on lower trophic levels of fish, are vulnerable to overfishing and therefore are good indicators of the level of fishing on a reef. | Serranidae | All groupers |
| | | Lutjanidae | *Aprion viriscens*<br>*Lutjanus bohar* |
| **Omnivores** | Second-level predators with highly mixed diets including small fish, invertebrates, and dead animals. Their abundance is a good indicator of fishing pressure | Haemulidae | All sweetlip |
| | | Lethrinidae | All emperor |
| | | Lutjanidae | All snapper except *Aprion viriscens & Lutjanus bohar* |
| **Corallivores** | Obligate and facultative corallivores are a secondary indicator of coral community health. | Chaetodontidae | Eight Butterflyfish: *C. bennetti, C. lineolatus, C. melannotus, C. meyeri, C. ornatissimus, C. trifascialis, C. trifasciatus, C. zanzibarensis* |
| **Invertivores** | Feed on coral competitors such as soft corals and sponges, and their abundance may be a secondary indicator of stability of these groups and of a phase shift. Also prey on small invertebrates in the benthos. | Pomacanthidae | Angelfish. All species except *Centropyge* spp. which are grazer-detritivores |
| | | Balistidae | Benthic triggerfish (e.g., *Sufflamen* spp.) |
| | | Chaetodontidae | Noncorallivore Butterflyfish: all other Chaeotdontids except *H. zoster* and *H. diphreutes* which are planktivore |
| **Planktivores** | Resident on reefs but feed in the water column. Their presence/absence may be related to water column conditions, suitable habitat for shelter or reef features such as passes. | Chaetodontidae | *Hemitaurichthys zoster,*<br>*Heniochus* spp. |
| | | Balistidae | *Triggerfish in the water column eg. Melichthys spp., Odonus niger* |
| | | Acanthuridae | *A. mata, A. nubilus, A. thompsoni, Paracanthurus hepatus All large Naso (>20 cm TL, 16 cm for N. hexacanthus), except Browsers (N. unicornis etc)* |
| | | Caesionidae | *All Fusiliers* |
| Detritivores | Feed on organic matter including diatoms in sediment and reef surfaces, high abundances poorly understood | Acanthuridae | *Ctenochaetus spp.* |
| Grazer-detritivores | Feed on algal turf and sediment to extract detritus, microbes and diatoms; may limit growth of macroalgae | Acanthuridae | *A. blochii, A. dussumieri, A. leucocheilus, A. nigricauda, A. xanthopterus, A. tennenti* |
| | | Pomacanthidae | *Centropyge spp.* |
| **Herbivores** | Feed on endolithic and epilithic algae, substratum, and macroalgae. Exert control on coral-algal dynamics, implicated in determining phase shifts from coral to algal dominance, for example, in response to mass coral mortality | | |
| Large excavators | Take few, large, deep bites, and remove calcareous substratum; play a large role in bioerosion | Scarinae | *Chlorurus spp. >35 cm, for example, C. strongylocephalos*<br>*Cetoscarus ocellatus*<br>*{Bolbometapon muricatum}* |
| Small excavators | Remove algae and substrate; play a smaller role in bioerosion | Scarinae | *Chlorurus spp. <35 cm* |
| Scrapers | Remove algae, sediment, and detritus by closely cropping or scraping the substrate | Scarinae | *Scarus spp., Hipposcarus spp* |
| Browsers | Feed on large macroalgae | Scarinae<br>Acanthuridae | *Calotomus* spp. *Leptoscarus* spp.<br>*Naso elegans, N. tuberosus, N. unicornis,* other *Naso spp.* <21 cm (<16 cm for *N. hecacanthus*) |
| Grazers | Graze epilithic algal turfs, including red algae; likely to limit growth of macroalgae | Acanthuridae | *Zebrasoma spp. A. nigrofuscus, other Acanthurus spp. for example, A. lineatus* |
| | | Siganidae | *Siganus spp.* |

NOTE: Trophic categories and feeding information based on (Choat and Clements 1998, Samoilys and Carlos 2000, Choat et al. 2002, Green et al. 2009, Clements et al. 2017).

## Readiness

Low to medium.

Fish size-frequency distribution is a useful indicator and can be determined from underwater visual census data. However it is most useful when individual species are identified, which requires skilled observers and is not always reported.

Size specific knowledge of demographic processes in reef fish is poorly described for a large proportion of coral reef fish species present on coral reefs (Froese & Pauly 2020).

## Methods and materials

Methods to describe fish size frequency are similar to those described for assessments of reef fish abundance, size and estimated biomass.

We recommend familiarization with training methods to establish consistent and accurate size estimations of reef fish during underwater visual censuses (English et al. 1997)

We further recommend the use of data mermaid (www.datamermaid.org) to enter data for belt transect surveys of reef fish, as this will facilitate accurate spelling of fish taxonomic names and will identify errors in recorded fish size based upon known size ranges in FishBase (Froese and Pauly 2020).

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## 19. CORALLIVORY

### Indicator Name

Corallivory

### Alternative indicator names

coral predation; feeding on coral

### Importance

High

### Definition

Corallivory is the predation on live corals, of the Order Scleractinia by other coral reef organisms such as fish and invertebrate predators which may consume the mucus, tissue, and skeleton of corals.

Corallivores are organisms that feed on scleractinian corals.

Quantification of the abundance, relative spatial distribution and feeding rates of corallivores, provides insight to the impacts on corals and the condition of a coral reef community.

### Background and Reasoning for choosing the indicator in Mozambique

The organisms that feed on live corals, impact coral growth and fitness as well as shape coral community dynamics. Corallivores are diverse and include a wide range of coral reef organisms that include fishes, echinoids, crustaceans, mollusks, and annelids. The feeding behaviors and strategies amongst coralivores that feed on established coral colonies are also diverse (Cole et al. 2008, Rotjan and Lewis 2008, Rice et al. 2019). These include:

- mucus feeders, such as crab of the genera *Trapezia* and *Tetralia*, that consume mostly coral mucus and some coral tissue,
- browsers that remove only coral tissue, such as butterfly fishes (e.g. *Chaetodon spp*.) and Crown-of-thorns starfish (*Acanthaster spp*.),
- scrapers such as parrotfishes, that scrape live tissue from the surface of corals and also remove a thin layer of skeleton as a consequence, and
- excavators, also referred to as bioeroders, that remove large amounts of coral skeleton when they remove coral tissue (e.g., pufferfishes and some parrotfishes).

Corallivores may feed on corals at any of the stages of a corals life cycle. Feeding on larvae or gametes during their planktonic phase is likely to be the least discriminatory, and perhaps not considered true corallivory, because corals are fed upon simply because they are present in the plankton (Hamner et al. 1988, Fabricius and Metzner 2004). Coral recruits may be targeted directly or a consequence of feeding on macroalgae (Rotjan et al. 2006, Mumby 2009, Trapon et al. 2013). Corallivory of adult colonies is reported to be more species specific, restricted to fewer coral genera than usually present on a reef, and preferences are shown by fish species for certain coral taxa (Burkepile et al. 2019), or for specific sections of the coral colony such as reproductively active tissues (Rotjan and Lewis 2009).

Corallivory frequently results in non-lethal impacts to corals, with consequences for the health and fitness for individual corals, and impacts on coral community dynamics. As coral colonies increase in size, corallivory is increasingly likely to result in partial mortality or minor lesions and not the death of the coral (Hughes and Jackson 1980, 1985, Hughes and Tanner 2000). However, predation on corals can inhibit coral growth and sexual reproduction through tissue and/or skeleton removal (Henry and Hart 2005). Corallivory may also act as a vector for coral diseases (Nicolet et al 2018).

Corallivory can range from minor feeding on individual polyps by butterfly fishes (family Chaetodontidae) to predation by crown-of-thorns starfish (*Acanthaster spp*.) that results in widespread coral mortality (Kayal et al. 2012, Saponari et al. 2015). Corallivory associated with outbreaks of COTS was the second largest cause of coral mortality on the Great Barrier Reef between 1985 and 2012 (De'ath et al. 2012).

Corallivory is a specialized feeding habit among marine species (Cole et al. 2008, Rotjan and Lewis 2008, Bonaldo and Rotjan 2018). At least 160 corallivores are known from worldwide studies on coral reefs, but corallivory is only reported to impact 28 genera of corals (Rotjan and Lewis 2008). There are 128 reported corallivorous fish species, less than five percent of  from 11 different families, (Cole et al. 2008). More than half (69) of these fish belong to the family Chaetodontidae, and between seven to ten species are reported from each of the families Labridae, Tetraodontidae, Balistidae, Monacanthidae, Pomacentridae and Scaridae. One third of coral-feeding fishes feed almost exclusively on corals, with more than 80% of their diet based on coral.
Corallivorous fish also show distinct prey preferences and consume only a small subset of available corals, most frequently of the genera *Acropora*, *Pocillopora* and *Porites* (Cole et al. 2008, Bonaldo and Rotjan 2018, Burkepile et al. 2019).

## This indicator is relevant to

Coral reef condition
Coral reef resilience
Coral diseases
Coral competition with corals, algae, sponges etc

## Benchmarks and thresholds

We recommend similar thresholds to those suggested for the partial mortality of corals (Indicator 14), given that the most immediate result of corallivory is the partial mortality of a coral colony.

Further to this, thresholds have been determined for the consideration of outbreaks of the crown-of-thorns starfish (*Acanthaster planci*) corallivore, with

≥ 1500 adult starfish km$^{-2}$ considered an outbreak that will lead to the degradation of a coral reef community (Moran and De'ath 1992). However, we caution that this threshold was determine nearly 3 decades ago, and recent studies provide indication that as a result of ocean acidification and climate change corals may have become increasingly susceptible to corallivory and bioerosion (Carballo et al. 2013, Fang et al. 2013, Rice et al. 2019).

## Readiness

Medium to low.

Assessing corallivory requires requires accurate identification of the species that predate on corals. If surveys , for example of reef fish, do not already indetify organisms to the level of species then key corallivores can be learnt and distinguished in surveys.

## Methods and materials

Corallivory can be assessed in two ways. First, by assessing the abundance of known corallivores at a reef. Methods for this have been described for Indicator 16 and 20. A second approach is to describe the frequency of feeding on corals, or rates of corallivory. The frequency or rate of corallivory can be determined by following individual corallivores and recording the frequency with which they feed on coral colonies, in the case of reef fish this is similar to describing bite rates of herbivores described for indicator 20. Alternatively individual coral colonies may be monitored with a stationary video camera to record all corallivores that feed on the coral within a period of time (e.g. 2-4 hours). Replicate video observations should be made for each coral species of interest. Video observations are likely to identify local corallivores, targeted coral species, and describe their relative importance of predators and prey providing detailed insight for Mozambique that may not be available in the international scientific literature. It may also be possible to monitor feeding scars and trends in partial mortality for corals associated with corallivory.

Metrics may be related to bite rates, percentage of coral surface eaten, number of feeding scars on a coral colony, abundance of corallivores in an area.

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## 20. HERBIVORY

### Indicator Name

Herbivory

### Alternative indicator names

Grazing, feeding or removal of algae or macroalgae, algal or macroalgal consumption

### Importance

Critical

### Definition

The term herbivory is used to refer to the reef organisms that feed on macroalgae, thus controlling and the biomass of macroalgae on a coral reef.

Rates of herbivory, refers to the quantified amounts of macroalgal removal or feeding on macroalgae per unit of time.

### Background and Reasoning for choosing the indicator in Mozambique

The ecological process of herbivory is key to the resilience of coral reefs (Bellwood et al. 2004, McClanahan et al. 2012), supports marine food webs (Adey and Goertemiller 1987, Adey 1998, Kramer et al. 2013), and maintains coral reefs with relatively abundant coral cover (Lewis 1986, Hughes 1994, Hughes et al. 1999, McCook 1999, Hughes et al. 2007a). Understanding the levels of herbivory is therefore an approach to understanding the state of a coral reef ecosystem (Littler and Littler 2007).

Macroalgae are important contributors to the dynamics of coral reef ecosystems (Littler and Littler 1994, 2013, Adey 1998, Fong and Paul 2011), and have diverse morphologies that range from diminutive filamentous algae and calcareous crusts to upright macrophytes (Steneck and Dethier 1994). Herbivores mediate the diversity and abundance of macroalgae, often shaping the condition of coral reef habitats (Lewis 1986, Hughes et al. 2007b, Burkepile and Hay 2010). Not surprisingly, the diversity of macroalgae is represented in the diversity of herbivores and their feeding strategies (Bellwood et al. 2004, 2019, Green et al. 2009).

The herbivores present in a coral reef ecosystems are diverse, including fishes, echinoderms, and molluscs, crustaceans and other micro-herbivore invertebrates, but they also differ in their feeding intensity and the spatial areas over which they forage (Carpenter 1986). Coral reef fish are frequently considered the most significant herbivores on coral reefs, but their feeding habits differ and result in variations in the removal of macroalgae and impacts on reef processes (Bellwood et al. 2004, 2019, Green et al. 2009).

Grazing by sea urchins can result in abrasion of coralline algae (O'Leary & McClanahan 2010) or coral recruits (O'Leary et al 2013). Coral reef fish, such as parrotfishes, can remove coral recruits with bites whilst grazing as a result of their small size (Mumby 2009), however fish herbivory is generally considered to benefit established corals and coral recruitment by widely reducing macroalgae competition on coral reefs (McCook et al. 2001, Hughes et al. 2007b, Birrell et al. 2008, Leary et al. 2013). Complementary feeding by diverse herbivores can produce positive, but indirect, effects on corals, and crustose coralline algae, although this unlikely under high abundances of sea urchins, which are both groups of key macroalgae for coral reef ecosystem (Burkepile and Hay 2008)

Grazing by reef fish plays a significant role in determining the state of a coral reef, but may also reflect the state of degradation of a coral reef (Littler and Littler 2007, Green et al. 2009). High density of macroalgae stands can deter herbivory (Hoey and Bellwood 2011). The traits of macroalgae determine the species of herbivorous fish that feed on them (Steneck and Dethier 1994, Bellwood et al. 2006). Greater abundance of macroalgae and lower coral abundance as well as the traits of macroalgae present result in differences in grazing on degraded versus recovering coral reefs (Chong-Seng et al. 2014). The presence of sediments in algal turfs deters fish grazing (Tebbett et al. 2017). Fish are more susceptible to predators in habitats with low coral cover and there they graze more quickly et  (Coker et al. 2014).

Herbivores may be subdivided into functional groups based upon "how" they feed on macroalgae as well as "what" impact they have on the community or ecosystem (

Table 1). This detailed understanding of the functional role of coral reef fishes is most well understood for herbivores and corallivores which have received substantially more research attention (Pratchett et al. 2018, Bellwood et al. 2019). Understanding herbivory from a functional point of view may be improved by also considering the influence of functional groups of macroalgae (Steneck and Dethier 1994). Samoilys et al. (2019) list key herbivores for the Southern Western Indian Ocean (

Table 2).

**Table 1. A detailed functional group matrix for coral reef fishes with an average total length >20 cm, that are roving herbivores present in the Indo-Pacific** (Bellwood et al. 2019).

| **'What' functional grouping or ecosystem function** | **'How' functional groupings** | **Description** | **Associated traits** | **Example species** |
|---|---|---|---|---|
| Brushers – remove particulates from the epilithic algal matrix but not algae | Brushers | Brush particulate material from algal turfs. | Bristle-like teeth, gizzard-like stomach | *Ctenochaetus striatus* |
| Grazers – remove algal material (<2 cm) from coral reefs | Croppers (Concealed) | Can nip-off short algae from concealed microhabitats. | Multi-denticulate teeth, elongated snouts, pairing behaviour | *Zebrasoma scopas, Siganus vulpinus* |
| | Croppers (sand 'suckers') | Remove algal material and particulates. | Multi-denticulate teeth, gizzard-like stomach | *Acanthurus olivaceus* |
| | Croppers (Open) | Nip-off short algae in exposed microhabitats. | Multi-denticulate teeth, thin-walled acidic stomachs | *Acanthurus lineatus* |
| | Scrapers | Scrape the outside of the reef matrix. | Beak like jaws with gracile associated musculature, pharyngeal mill | *Scarus rivulatus* |
| | Excavators | Bite into the reef matrix. | Beak like jaws with robust associated musculature, pharyngeal mill | *Bolbometopon muricatum* |
| Browsers – remove algal material (>2 cm) from coral reefs | Chopper | Remove the entire algal thallus. | Rod/peg-like teeth, robust adductor mandibulae muscle | *Kyphosus vaigiensis, Naso unicornis* |
| | Shearer | Only nibble on leafs. | Bicuspid teeth, small gape, light adductor mandibulae | *Siganus canaliculatus* |

**Table 2. Key herbivores of the Southern Western Indian Ocean** (Samoilys et al. 2019). **Herbivores are defined as feeding on endolithic and epilithic algae, substratum, and macroalgae; Exerting control on coral-algal dynamics, implicated in determining phase shifts from coral to algal dominance, for example, in response to mass coral mortality.**

| **Herbivore sub-grouping** | **Description of feeding habits and reasons for selection** | **Group/ family** | **English name or species** |
|---|---|---|---|
| Large excavators | Take few, large, deep bites, and remove calcareous substratum; play a large role in bioerosion | Scarinae | *Chlorurus spp. >35 cm, for example, C. strongylocephalos*<br>*Cetoscarus ocellatus*<br>*{Bolbometapon muricatum}* |
| Small excavators | Remove algae and substrate; play a smaller role in bioerosion | Scarinae | *Chlorurus spp. <35 cm* |
| Scrapers | Remove algae, sediment, and detritus by closely cropping or scraping the substrate | Scarinae | *Scarus spp., Hipposcarus spp* |
| Browsers | Feed on large macroalgae | Scarinae | *Calotomus* spp.<br>*Leptoscarus* spp. |
| | | Acanthuridae | *Naso elegans, N. tuberosus, N. unicornis,* other *Naso spp.* <21 cm (<16 cm for *N. hecacanthus*) |
| Grazers | Graze epilithic algal turfs, including red algae; likely to limit growth of macroalgae | Acanthuridae | *Zebrasoma spp. A. nigrofuscus, other Acanthurus spp. for example, A. lineatus* |
| | | Siganidae | *Siganus spp.* |

## This indicator is relevant to

Coral reef condition
Coral reef resilience
Macroalgae phase shifts
Coral reef degradation

## Benchmarks and thresholds

The thresholds identified for overall fish biomass on coral reefs may be applied, whilst more specific thresholds for herbivores are more challenging to identify.

In Fiji, within reserve reefs, representing good condition, total herbivore biomass of ~ 2 to ~ 8 kg per 100m$^2$, whilst in non-reserve reefs which represent poor condition total herbivore biomass ranged between ~ 0.27 to ~ 0.43 kg per $100m^2$ (Rasher et al. 2013).

In American Samoa total herbivore biomass ranged from ~ 300 g 100 m2 to ~ 1900 g/ 100 m2 for a variety of coral reef conditions, and higher herbivore biomass was associated with better coral reef condition (e.g. lower macroalgae cover and higher coral cover) (Heenan and Williams 2013).

On the Great Barrier Reef total herbivore biomass ranged from 0.8–7.9 kg per 100 $m^2$ on the inner shelf to 5.0–31.7 kg per 100 $m^2$ on the outer shelf which are generally associated with better condition and lower macroalgae cover (Wismer et al. 2009).

Reported from East African reefs, including Mozambique, the biomass of herbivore fish families Acanthuridae ranged from approximately 0.19 kg per 100 $m^2$ to 3.2 kg per 100 $m^2$, the biomass of Scaridae ranged from approximately 0.06 kg to 1.9 kg per 100 $m^2$ and the biomass of Siganidae ranged from approximately 0.03 kg per 100 $m^2$ to 0.5 kg per 100 $m^2$ (McClanahan 2019).

Several studies report that approximately half of the fish herbivore biomass on coral reefs is of the grazer/detritivore functional group, with browsers often representing ~ 5 % of the herbivore biomass (Heenan and Williams 2013, Rasher et al. 2013).

Sea urchin abundances in northern Mozambique are reported to range from 3.0 (± 3.0) per 10 $m^2$ to 29.1 (± 16.6) per 10 $m^2$ (McClanahan and Muthiga 2017).

We suggest ranges for herbivore biomass in Mozambique associated with different coral reef condition can be determined from local surveys.

### Readiness

Low to medium.

Detailed interpretation of the nature and levels of herbivory in a coral reef ecosystem requires detailed information for the herbivores present, their abundance and the intensity of their grazing. This is normally restricted to herbivorous fish and is best achieved when fish are identified to the level of species or at least genera during surveys. Such detailed fish surveys require experienced and skilled observers, although this requirement may be lightened by use of videos for surveys, which enable more careful analysis on land.

### Methods and materials

Herbivorous fish can be surveyed as part of general surveys of coral reef fish (English et al. 1997, Samoilys and Carlos 2000, Labrosse et al. 2002, Hill and Wilkinson 2004). Detailed identification of the species present is recommended to enable separation of the data into herbivore functional groups. Please consult methods described for fish abundance and fish functional groups (Indicator 16).

Surveys of herbivorous fish should be undertaken at similar times of the year and in comparable habitat to account for natural variation in herbivore presence and activity as well as macroalgae seasonality. Herbivores are more active on reef slopes than other reef habitats such as reef flats, lagoons or deeper reefs (Hay et al 1984). The community composition of herbivorous fish is also spatially variable, reflecting reef habitats together with the abundance of herbivores (Hay 1981, 1984a, Lewis and Wainwright 1985). Macroalgae condition and biomass is seasonal, and herbivores are likely to vary what they feed on to reflect food availability (Hay et al. 1983, Diaz-Pulido et al. 2009, Lefèvre and Bellwood 2010, 2011). Therefore, the presence and feeding intensity of herbivores varies seasonally (Lefèvre and Bellwood 2011, Afeworki et al. 2013b, 2013a, Wilson et al. 2014).

Two approaches to determine grazing intensity have frequently been used: grazing assays, and surveys of bite rates. Grazing assays are undertaken by placing pieces of macroalgae or seagrass food of known weight and or size in a coral reef habitat for a specified amount of time, an afterwards it is determined how much was consumed by herbivores (Hay 1981). Considerations for the choice of food include that it is a species available in the coral reef ecosystem being studies, and also that the macroalgae is sourced form a representative habitat for the study, because macroalgae defenses and feeding preference of herbivores differ between macroalgae source habitats (Hay 1984a).

Grazing intensity is also assessed by following individuals of a species of herbivore and counting the number of bites taken on macroalgae in a length of time to determine bite rates (McClanahan et al. 1994, Fox and Bellwood 2007, Afeworki et al. 2013b). Following reef fish for several minutes (e.g. 3 minutes) has been a common approach, but it must be recognized that human presence influences fish behaviour and there are biases associated with these methods (Dickens et al. 2011). However if the focus is changed from the individual fish to a specified area of reef substrate, such as a quadrat, then a stationary video offers a suitable alternative to record grazing rates of multiple species simultaneously.

Sea urchins are also important herbivores that compete with fish and influence the structure of coral reef ecosystems (Steneck 2013, 2020). The abundance of sea urchins can be surveyed in determined areas of reef such as circular areas, quadrats or belt transects (McClanahan 1998). Sea urchin grazing rates may also be determined with feeding assays and distinguished by characteristic bite marks or with the use of video (Hay 1981, 1984b). As with reef fish it is important to recognize that the abundance and community composition of sea urchins is habitat specific (Coppard and Campbell 2005).

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# 21. CORAL REEF COMPLEXITY

## Indicator Name

Coral reef complexity

## Alternative indicator names

3D Structure

## Importance

High

## Definition

The complexity of a coral reef refers to the physical three-dimensional form, a result of vertical relief and variety of structure present in a coral reef habitat.

## Background and Reasoning for choosing the indicator in Mozambique

Complex three-dimensional structure of a coral reef provides more habitat niches that can support greater diversity and biomass of fish and other living organism in the reef community (Wilson et al. 2006, Graham and Nash 2013). Three dimensional complexity supports high species richness found on coral reefs (Graham et al. 2007, 2015, Newman et al. 2015). There is also a link between reef structural complexity and the ability recover from degraded states (Rogers et al. 2015). Greater structural complexity can also promote habitat diversity that favours coral recruitment following disturbances (Gouezo et al. 2020). Therefore, maintaining reef complexity improves the condition of a coral reef and can increase reef resilience in the long term.

The underlying three dimensional structure of a coral reef is determined by bathymetric settings, past reef morphology, geological processes, and past physical disturbance events (Chappell 1974, Scoffin 1992, 1993, Woodroffe 1992, Ota et al. 1993, Kennedy and Woodroffe 2002). Coral reef accrete on past structure over time as a result of framework growth, physical and biological erosion and internal sedimentation and cementation (Scoffin 1992, Rasser and Riegl 2002, Perry and Hepburn 2008)

Growth of corals contributes to building new carbonate skeletons that accretes coral reef structure (Chappell 1980) together with calcareous material produced by algae (Adey 1975, 1998, Chisholm 2000, 2003). Rubble and loose carbonate material is temporarily cemented by organisms such as crustose algae (Adey 1998, Chisholm 2000, Weiss and Martindale 2017) and sponges (Wulff 1984). Living organisms continuously contribute to the breakdown of reef structure through bio-erosion, creating crevices and loose material within the reef substrate and matrix (Hutchings 2000, Schönberg 2008, Carballo et al. 2013, Glynn and Manzello D.P. 2015, Bak 2019). Physical breakdown and erosion also occurs as a result of storms, wave action and water flow (Harmelin-Vivien 1994, Torda et al. 2018). Chemical erosion also degrades coral reef structure (Perry and Hepburn 2008) and increasingly threatens coral reefs under changing climatic conditions and ocean acidification (Pandolfi et al. 2012).

Provided the three dimensional structure of a reef remains, the loss of live coral cover may have few short term and direct effects on species of fish and invertebrates, except for those that feed directly on the coral or live in live coral structures (Pratchett et al. 2018). However, without live corals to maintain the reef structural complexity, degradation and erosion of the reef structure occurs and impacts reef fish and other organisms (Wilson et al. 2006, Graham and Nash 2013, Pratchett et al. 2018). This flattening of a coral reef structure is widely associated with the degradation of a coral reef ecosystem and the ecosystem services it provides (Alvarez-Filip et al. 2009, Newman et al. 2015, Darling et al. 2017).

Physical drivers of habitat zonation on coral reefs define the levels of complexity that can be maintained on a coral reef by influencing the morphology of the corals present. This occurs as a result of the influence of wave action, light, sedimentation (Chappell 1980, Geister 1981, Graus and Macintyre 1989). This implies that comparisons of coral reef complexity as a measure of coral reef condition should be undertaken between similar habitats.

## This indicator is relevant to

Coral reef condition
Coral reef resilience
Coral reef fish communities
Coral reef fisheries

## Benchmarks and thresholds

Habitat zonation plays a significant role in determining the complexity of a coral reef making it challenging to determine a general threshold for coral reefs. Decreases in coral reef complexity should be interpreted to represent a decline in coral reef condition. To this end it has been suggested that a decline of ≥8% in rugosity between successive surveys should represent a decline in coral reef condition (Alvarez-Filip et al. 2011, Flower et al. 2017).

## Readiness

Rugosity is easily measured with basic field methods and can be undertaken parallel to field surveys of corals and other benthic organisms or reef fish.

More complex measurements of rugosity requiring three dimensional imagery or LIDAR scans requires the dedicated work of specialized personnel.

## Methods and materials

The rugosity of a coral reef is a measure of reef complexity frequently undertaken with a chain (or sinking rope) of known length that is fitted to the contours of the substrate along the line of a taught transect tape (Luckhurst and Luckhurst 1978). The shorter the distance the chain extends along the transect tape the greater the reef rugosity. Thus, reef rugosity can be measured using a 10 m long, fine-linked chain at the start of the transect. The chain should be carefully fitted to the substrate contour, hugging crevices and structures, along the initial section of a 50 m transect also sued for surveying corals and fish communities. The linear distance to the point where the chain fitted to the substrate ended was recorded from the transect tape. To calculate a value for rugosity (R), the linear distance of the fitted chain in cm is subtracted from 1000 (the length of chain in cm).

Rugosity = linear length of fitted chain (cm) – length of chain laid flat (cm)

Reef complexity can also be estimated quantitatively with visual categorisation of reef complexity. Macro-complexity can be classified at a site wide level by noting the overall structure of each site at intervals of no more than 5 m along each transect (e.g. PIT or fish belt transect) and classifying the overall structure of a site according to 6 categories (Figure 1). Micro-complexity may be described at every point of observation along a PIT transect with reference to 5 point scale (Table 1) (Wilson et al. 2007).

**Figure 1. Categories 0 to 5 of macrocomplexity used to describe the habitat at each site. Categories are 0) no vertical relief, 1) low, widespread relief, 2) low to occasionally moderate relief 3) consistent moderate relief, 4) complex vertical relief, 5) fissures, caves, overhangs**

**Table 1. Categories of microcomplexity used to describe point along each PIT**

| Microcomplexity | category |
|---|---|
| Totally flat (e.g. sand) | 1 |
| Rubble; small patches; minimal relief | 2 |
| Mounding; medium structure | 3 |
| Sub-massive or coarse branching | 4 |
| Branching, complex and crevices | 5 |

Modern techniques have also been explored to undertake detailed descriptions of the complexity of a coral reef substrate. These include LIDAR (Brock et al. 2004) and three dimensional imagery (González-Rivero et al. 2014), which can be collected over relatively large spatial scales, but which require greater technical capacity and are more costly. Photogrammetry measurements can be used to construct 3D models of coral colonies or for more precise measurements. Suggested software for these purposes include EventMeasure (www.seagis.com.au) and Agisoft Metashape (www.agisoft.com). Precise volume calculations of colonies modeled at different points in time are made possible by the photogrammetric 3D model approach. The disadvantages include higher costs for equipment and software, as well as more time required underwater, however photogrammetry is a precise and non-intrusive method for measuring coral growth. Not yet widespread, the technology for these approaches is advancing rapidly and benefiting from increasingly available technologies and software that allow for more efficient workflow (Burns et al. 2015, 2016, Raoult et al. 2016, House et al. 2018)

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# 22. COVER OF RUBBLE AND SAND

## Indicator Name

Cover of rubble and sand

## Alternative indicator names

Unconsolidated substrate, loose substrate

## Importance

High

## Definition

The proportion of the reef substrate that is covered by unconsolidated rubble or sand.

Rubble on coral reefs can be defined as dead coral skeleton or reef rock pieces that have fractured and been liberated by mechanical or chemical means and are larger in size fraction than sand (>2mm) (Rasser and Riegl 2002)

## Background and Reasoning for choosing the indicator in Mozambique

The presence of unconsolidated substrate in a coral reef ecosystem provides insight to the condition of the reef. Unconsolidated substrate hinders the attachment of recruits and survival of sessile organisms, such as corals (Rogers 1990, Demartini et al. 2013, Trapon et al. 2013, Chong-Seng et al. 2014, Moeller et al. 2017, Speare et al. 2019). For this reason high levels of unconsolidated sand and rubble can reduce the establishment of benthic communities. The larger the unconsolidated material the more likely that organisms such as sponges, corals and crustose coralline algae can temporarily consolidate it (Wulff 1984, Adey 1998), later allowing for more permanent consolidation to occur (Scoffin 1992, Perry and Hepburn 2008)

It is natural to have patches of unconsolidated material in coral reef ecosystem, representing a mosaic of natural process of disturbance and erosion and successional stages of a reef community (Dollar and Tribble 1993, Rogers 1993, Connell et al. 1997). Nonetheless, increasing levels of rubble or sand may provide indication of degradation of the coral reef ecosystem and it is important to assess whether this results from natural processes or human influence.

Rubble and sand naturally result from the erosion of carbonate material on coral reefs. This may result from the bio-erosion of organisms such as hard corals or calcareous algae and reef matrix (Hutchings 2000, Perry and Hepburn 2008) as well as the fragmentations of organisms during storms (Rogers 1993, Scoffin 1993, Lirman and Fong 1997). The breakdown of carbonate material, particularly coral skeletons, is greater on degraded reefs and after mass coral mortality resulting from bleaching, crown-of-thorns starfish outbreaks, disease or terrestrial runoff and pollution (Holmes et al. 2000, Pandolfi et al. 2005, Lasagna et al. 2010, De'ath et al. 2012, Carballo et al. 2013, S A et al. 2018)

Human action also influences the amounts of sand and rubble in a coral reef environment. Rubble can result from any physical disturbance, which may range from ship groundings (Negri et al. 2002, Rubin et al. 2008) anchoring (Forrester 2020) and dynamite fishing (Fox 2004, Braulik et al. 2015) to trampling by gleaning fishers or tourists snorkeling and reef walking (Andréfouët et al. 2013). The abundance of sand and sediment on a reef may be influenced by dredging, artificial beaches or alterations to water circulation and terrestrial runoff (Moeller et al. 2017)

Restoration interventions may be undertaken to stabilize unnaturally high amounts of unconsolidated substrate. This has been done for example in response to coral reef degradation resulting from dynamite fishing (Fox 2004, Fox et al. 2019, Williams et al. 2019) or ship groundings (Ceccarelli et al. 2020). Any interventions should seek to understand whether the levels of sand and rubble are unnaturally high in the first place, whether this is hindering natural recovery and the ecological function of the reef community, and should also consider the long term dynamics of the reef ecosystem (Ceccarelli et al. 2020)

It is important to recognize that rubble and sand patches in a balanced environment represent unique habitats in the coral reef ecosystem that contribute to biodiversity and ecosystem function. Patches of rubble and sand offer habitat for many free living corals (Veron and Stafford-Smith 2000), many damselfishes, crustaceans, molluscs and cryptic species (Scoffin and Tudhope 1985, Wilson et al. 2008, Kramer et al. 2014). Concerns relate to indications that increasing degradation of coral reef ecosystems worldwide is contributing to higher than normal erosion of coral reefs (Alvarez-Filip et al. 2009, Cheal et al. 2017). Ocean acidification may already have contributed to weaker coral skeletons and higher rates of their breakdown into rubble and sand (Madin et al. 2008, 2012). All of which has long term deleterious impacts on the three dimensional structure, biodiversity and biomass of reef fish and invertebrate communities on coral reefs (Newman et al. 2015, Pratchett et al. 2018).

## This indicator is relevant to

Coral reef condition
Coral reef degradation
Geological processes of coral reef accretion
Coral reef diversity
Coral recruitment

## Benchmarks and thresholds

Initial surveys of a coral reef should be undertaken to establish reference levels for the cover of sand and rubble. Monitoring can compare the trajectory of a reef against this reference to determine whether reef condition has improved (sand

and rubble decline) or degraded (sand and rubble increase). Nearby reefs may be used as a contextual reference, although it is important to realize that the individual history and settings of each reef impact levels of sand and rubble.

### Readiness

High

The cover of rubble and sand in coral reef ecosystems can easily be assessed simultaneously to other benthic organisms and requires basics skills and experience.

### Methods and materials

Cover of rubble and sand should be recorded separately and can be measured as a percentage of the substrate. Similar methods to assessing the benthic cover of sessile organisms (see indicator 1).

The cover of rubble and sand can also be assessed with relative success using remote sensing or aerial imagery (Li et al. 2020).

Greater detail in the description of the quality of unconsolidated material can be undertaken using standard geological classifications of grain size (Scoffin and Tudhope 1985, Scoffin 1992, Perry and Hepburn 2008, Tebbett et al. 2017).

Unconsolidated surfaces, in particular rubble, may be partially covered by sponges, coral recruits and crustose coralline algae amongst other benthic organisms reflecting a natural successional development within a coral reef ecosystem. It is recommended that surveys clearly define such occurrences. This is reflective of the scale of observations and for example surveys using transects, belt transect or quadrats (English et al. 1997, Hill and Wilkinson 2004) should define the substrate as ruble and may make secondary observations to distinguish points with organisms settled onto the rubble. Quantitative assessments of the level of unconsolidated substrate should seek to identify stability or change, as either increases or decreases in the levels of unconsolidated substrate.

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