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Supplementary information

Vulnerability to collapse of coral reef ecosystems in the Western Indian Ocean

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Vulnerability to collapse of coral reef ecosystems in the Western Indian Ocean

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1. Introduction

This Supplementary Material to the publication "Vulnerability to collapse of coral reef ecosystems in the Western Indian Ocean" by D. Obura and colleagues. It documents all details of the Red List of Ecosystems assessment of Western Indian Ocean coral reefs. It should be read in conjunction with the main paper, as well as with the key IUCN RLE publications, in particular the *Guidelines for the application of IUCN Red List of ecosystems categories and criteria* [1].

This document mirrors the technical submission for this Red List of Ecosystems assessment to the IUCN RLE Unit for approval (available at https://iucnrle.org/assessments/).

2. Ecosystem definition

2.1. Area of assessment

We focused our study on photic coral reefs, an Ecosystem Functional Group at level 3 of the IUCN Global Ecosystem Typology [2], within the Western Indian Ocean. The Western Indian Ocean (WIO) is a Province under the Marine Ecoregions of the World (MEOW) [3] and corresponds to the UNEP Regional Seas region extending from Kenya to South Africa on the East African coast, and the adjacent islands of the southwest Indian Ocean (fig. 1a, fig. S1).

We identified 11 ecoregions for analysis (Table S1), with the following characteristics:

- To match existing ecoregion descriptions as closely as possible [3,4];
- Recognise separation of Comoros from mainland Eastern Africa and Madagascar based on corals [5] and fish [6]
- To match nationally-determined ecoregions, to maximize future use in planning and implementation processes, as in Madagascar [7,8];
- Limit the Kenya/northern Tanzania ecoregion at the Kenya-Somali border due to lack of data from reefs north of this point.
- Allow for a gap between the Delagoa/southern Mozambique and the northern Mozambique/western NMC ecoregion due to the lack of coral habitats due to the Zambezi River (the Sofala ecoregion of [3]).

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- Ecoregions with little data were combined to improve analyses and completeness of this study, but staying true to boundaries defined in other processes – e.g. combining two northern ecoregions in Madagascar (NE and NW Madagascar), and two ecoregions across the Seychelles;

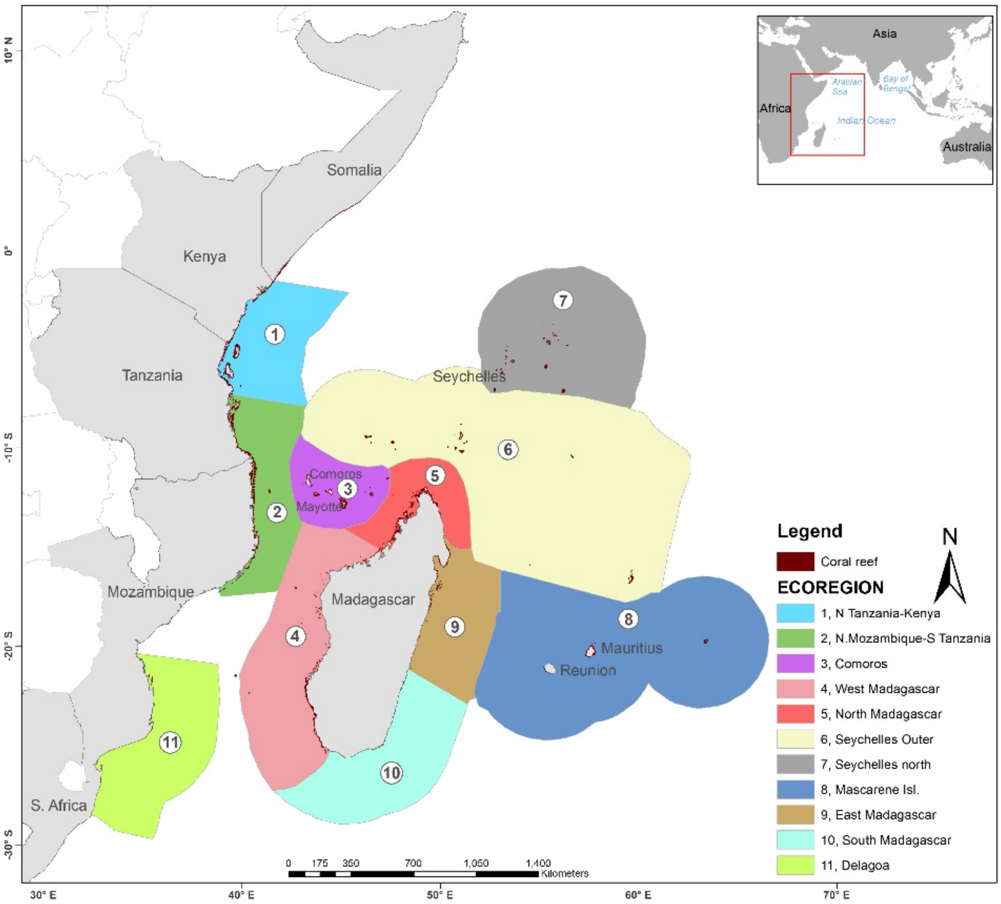

Figure S1. Ecoregions used in the Western Indian Ocean Red List of coral reef ecosystems assessment.

The geographic coverage of coral reefs was obtained from the Millennium coral reef layer [9] curated by the World Conservation Monitoring Centre (https://data.unep-wcmc.org/datasets/1), enabling calculation of reef area (Table S1), with some corrections applied recently by [10] and a localized correction for Delagoa provided by the Oceanographic Research Institute of South Africa (Source: Sean Porter).

Table S1. Ecoregions assessed within the Western Indian Ocean. Sources cited in the table include (1) Spalding et al. 2007, (2) Obura 2012, (3) Samoilys et al. 2019 and (4) IUCN (unpublished). Coral reef area calculations were obtained from analysis of the Millennium coral reef data layer available at https://data.unep-wcmc.org/datasets/1.

Ecoregion		Ecoregion notes	Coral reef area (km^2)
1	N.Tanzania-Kenya	Most closely follows (2) – N Tanzania/Kenya/ Monsoon Coast) though with no data from southern Somalia.	1778
2	N.Mozambique-S.Tanzania	The Tanzania/Mozambique continental coast in the East African Coral Coast (1) and Northern Mozambique Channel (2).	3115
3	Comoros	Comorian archipelago within the Northern Mozambique Channel (2), distinguished by fish fauna (3).	888

4	West Madagascar	West coast of Madagascar in the Mozambique Channel, with boundaries determined by (4), to support national processes.	1428
5	North Madagascar	NE and NW Madagascar are separated in (4) but low number of study sites in each, so they were joined, in agreement with (1) and (2). Outer boundaries as in (4), to support national processes.	1278
6	Seychelles Outer	Seychelles ecoregions distinguished more by data availability, separating the northern granitic island on the Seychelles bank and more populated/visited islands, from the more remote outer islands including the isolated banks (Saya de Malha, Nazareth and Cargados Carajos)	1147
7	Seychelles North		1041
8	Mascarene Islands	The remote south western islands of Reunion, Mauritius, Rodrigues (as in (1) and (2)) and Tromelin	602
9	East Madagascar	East Madagascar coast using the boundaries of (4) to support national processes, though (1) and (2) advocate different boundaries.	504
10	South Madagascar	The 'Grand Sud' of Madagascar as in (2), is a temperature enclave influenced by cool upwelling [11]. Marginal coral reef environment.	31
11	Delagoa	Matching the boundaries of (1) including all coral reefs of southern Mozambique and northern South Africa.	107
WIO region			11,919

Coral reef structure in the WIO is predominantly made up of fringing reefs around islands and continental coasts, with narrow lagoons, with some atoll structures in the Seychelles [12,13]. While a wider diversity of coral genera is present on fore reef slopes benthic cover is relatively similar on fore reefs and in lagoons. Mirroring other regional analysis in the WIO, and insufficient sampling across reef zones to allow regional comparisons, we lump data from all reef zones together [14,15].

2.2. Abiotic environment

Coral reefs are biogenic structures limited to warm, shallow (rarely >60 m depth), clear, relatively nutrient-poor, open coastal waters, where salinity is close to full, alkalinity is high and water temperatures vary between 17-34°C. Outside of these parameters the symbiosis between corals and zooxanthellae is compromised and unable to fix sufficient energy from sunlight to facilitate the skeletal development that builds up the reef framework over time. In the WIO, accreting coral reef communities tend to occur in < 30 m of water on the mainland and Madagascar coasts due to turbidity from terrestrial runoff, and deeper on the small islands. The primary envelope for corals is shifting with climate change as ocean waters are warming (≈ 0.8 C since pre-industrial) and acidifying (decrease of 0.1 pH) [16]. Warming conditions drive corals closer to their upper thermal tolerance thresholds, and acidifying seawater imposes a metabolic cost on corals and dissolves reef frameworks faster [17]. Thus climate change poses a fundamental existential threat to coral reefs, documented not just in recent impacts to corals through regional and global mass bleaching events [18], but also in projections of future climatic conditions [16].

2.3. Characteristic native biota

Coral reefs are biogenic structures built up through calcium carbonate deposition enabled by the symbiosis between scleractinian corals and endosymbiotic dinoflagellate zooxanthellae [2,19]. Calcium carbonate accreted by coral colonies and other components of the reef ecosystem over decadal and longer time-scales builds up the geomorphological structure of the reef. The complex 3-dimensional structure provides a high diversity of niches and resources that support a highly diverse biota, while high light intensities and water movement on shallow tropical shorelines provide a rich environment supporting this diversity and highly productive trophic webs. Coral reefs are characterized by typical ecological functions and processes (Section 2.5), delivered by a wide variety of species due to the high diversity within reef ecosystems, and across different geographic locations and biogeographic zones. As a result, coral reefs demonstrate high levels of functional redundancy [20].

Coral reefs of the WIO have been described extensively [12] and most recently their status updated in regional status reports [15,21]. The hard coral fauna of the region comprises some 350-400 species [4,5], of which 90% are Indo-Pacific in distribution, 5% are restricted to the western and northern Indian Ocean including the Red Sea and gulfs, and the final 5% restricted mainly to the Western Indian Ocean, a result of tectonic movements and connectivity over the last 30 million years [22]. Soft corals are also a dominant benthic fauna, though less studied, particularly ground-covering leathery forms that compete with hard corals, particularly in high-sediment and wave energy zones. The WIO as a distinct Province within the Indo-Pacific realm supports just over 2,400 fish species [23]. There are over 3,000 tropical reef fish species found in the Indian Ocean of which 74% (2,383) range widely through the Indo-Pacific leaving ~ 850 endemic to the Indian Ocean [24]. The WIO is well established as a second peak in fish diversity after the Coral Triangle [25]. The highest coral reef fish species richness in the Indian Ocean is found in the west, with ~ 600 to 960 species [24] on the east African continental coastline. The highest level of endemism in the WIO is found in the east, in the Mascarene Islands of Reunion and Mauritius [26], which is typical of peripheral regions in the Indo-Pacific [23,27].

2.4. Threats

Coral reef ecosystems in the WIO face a diversity of threats from local to larger scales, now extending to the global [13]. Natural threats or disturbances include damage from waves during storms and cyclones, sedimentation due to flooding from seasonal and extreme rains, via river mouths or direct runoff from land. The threat from sedimentation is exacerbated by anthropogenic factors, including land use change that results in greater runoff and erosion of soils, and coastal development that changes water flow patterns, including of groundwater. Coral reefs provide a number of ecosystem services that people benefit from, and access to these resources induces some level of damage – particularly extraction of fish and other organisms for food and other purposes, direct damage by tourism and recreation, and attraction of people to coral reef coastlines because of these benefits and the shoreline protection offered by reefs. In the Western Indian Ocean fishing is pervasive, with high levels of dependence of small scale and artisanal fishers [28]. A result is depleted fish populations in all but the most remote and effectively protected locations [6,29]. Sedimentation and pollution impacts at river mouths and points of coastal urban development are increasing (Obura 2015). Cyclone damage to reefs in the WIO is limited to some outer islands (in the Seychelles and Mascarene islands) and the Madagascar coast > 10° S. Coral predators (such as crown of thorns) and diseases are not widely prevalent, and only reported to have sporadic localized impacts [30].

Global threats affecting reefs in the WIO include warming, with the first global bleaching event in 1998 [31] being the most significant yet, but with a host of smaller regional and two more global bleaching events since [14,21]. Recent estimates were that the 1998 bleaching event reduced coral cover by a step change of 30%, after which recovery from impacted reefs was balanced by degradation of others, and the 2016 event (the second largest) reduced coral cover by a further 20-25% [15]. The impact of acidification has not yet been quantified in the WIO.

Based on the above, and data limitations on abiotic factors, we assessed mass mortality of corals due to thermal stress and fishing, as the primary threats affecting coral reefs across the WIO. Thermal stress was assessed from primary threat data in Criterion C (section 5), while the threat from fishing was assessed by inference from grouper populations in Criterion D (section 6).

2.5. Coral reef ecosystem model

The coral reef ecosystem model (fig. 2, S2) we developed focused on four main compartments: hard corals, fleshy algae, herbivorous fish and piscivorous fish. Hard corals are primary calcifiers on coral reefs, classed as 'ecosystem engineers' [32]. Algae represent the dominant alternative cover group to corals on coral reefs, with turf, fleshy and calcareous algae having competitive roles alongside non-calcifying invertebrates [33,34]. We focused on two key functions of fish – herbivory, that controls coral-algal dynamics and thus the dominance of calcifying versus non-calcifying taxa [35–37], and piscivory, which exerts top-down control on trophic webs on the reef [20,38,39].

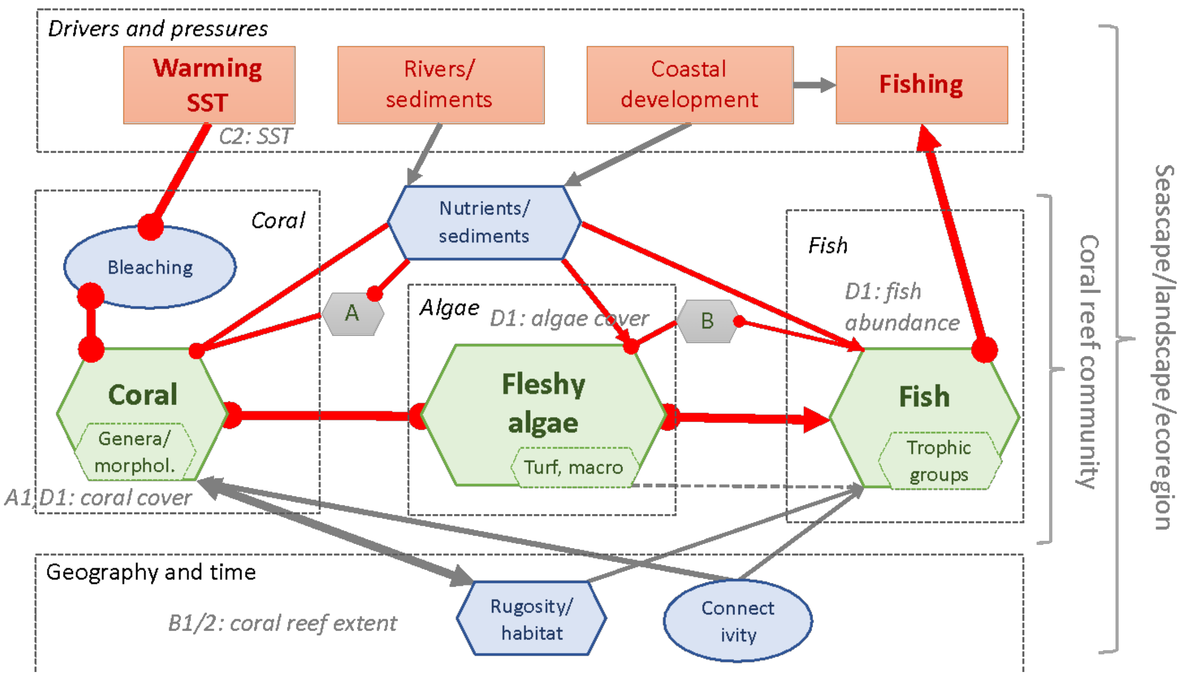

Figure S2. General coral reef ecosystem model applied in assessing the risk of collapse for Western Indian Ocean coral reefs. Compartments assessed in this study include corals, fleshy algae and fish (parrotfish and groupers), with pressures from warming SST and fishing, as shown in fig. 2. Sub-compartments which may be assessed in other regions might include coral diseases or predators (which would be placed in Compartment A) and other herbivores such as sea urchins (Compartment B). Model symbols follow the RLE guidelines: symbols (e.g. green hexagons - major ecosystem components; blue ellipses - ecosystem processes; blue rectangles - structural features, orange rectangles - drivers and pressures) [1].

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The broader model may include key interactions relevant to, but of secondary importance, in reef dynamics, for example, of sea urchins and other macroconsumers that may control algae populations, and of coral predators and coral diseases that may control coral populations (fig. S2). These may in turn be affected by nutrients and water quality factors as well as by other ecological interactions such as predation by fish. Additional phenomena included in the general reef model include coral bleaching and connectivity, which have both biological and physical elements, nutrient/sediment interactions and rugosity/habitat characteristics. Anthropogenic drivers and pressures strongly impact on coral reef ecosystem dynamics in multiple ways, but in particular through climate-related warming, fishing and other extraction, and runoff from rivers and land, which itself is affected by coastal and economic development. Given data constraints and for parsimony, not all of these interactions were or could be quantified in this RLE assessment, and we focus on the core compartments of corals, fleshy algae and two trophic groups of fish – herbivores, represented by parrotfish, and piscivores, represented by groupers – as documented in fig. 2.

2.5.1. Parametrizing the model

In this implementation of the RLE we were able to obtain datasets allowing quantification of the following aspects of the model:

- Corals – percent hard coral cover;
- Algae – percent fleshy algae cover, as the sum of turf algae, macroalgae and calcareous algae (e.g. *Halimeda*), when available. Note that some programmes use 'fleshy' and 'macro' as synonyms, but here, 'fleshy' algae is a broader group than macroalgae, in agreement with emerging usage in the GCRMN and for consistency with EOV definitions for coral reefs [40,41];
- Fish – abundance of parrotfish and groupers, as representatives of herbivorous and piscivorous fish, respectively;
- Warming SST/coral bleaching – projection of future sea surface temperature;
- Fishing impacts – abundance of groupers, as particularly vulnerable to fishing, but also parrotfish.

2.6. Collapse definition

The categories for assessing risk of ecosystem collapse have been derived from those for species extinction, covering 8 categories from Not Evaluated (NE) to Collapsed (CO) (fig. 1).

A key challenge in assessing the risk of collapse of an ecosystem, particularly a complex one such as a coral reef, is answering the question 'what constitutes collapse?' The RLE method defines ecosystem collapse as the absence of key biota and interactions characteristic of the ecosystem, and acknowledges a number of challenges with doing this in sections 3.2 [1] and 'Ecosystem Collapse and Risk Assessment' [42] and through further testing [43]. Identifying the point of collapse is operationalized by identifying key variables quantifying the key biota and interactions, and setting thresholds below which they can be said to be functionally absent (Table 1).

For coral reefs, the absence of hard corals is a primary indicator of coral reef collapse, but identifying an operational minimum threshold of coral cover below which their function in creating the reef ecosystem is lost requires addressing assumptions. At the same time, thresholds of collapse for other compartments, here algae and fish, need to be determined, with similar caveats. And given there are multiple compartments to the model, must all of them be below collapse thresholds for the system to be collapsed, or just one, or somewhere in between? Current RLE practice assigns the highest risk category across indicators within and across criteria to the overall ecosystem risk , however in complex ecosystems with multiple compartments and interactions of different hierarchy and strength, this may not apply. For example, in the Meso-American reef study [44] this problem did not arise as both coral and piscivore compartments were rated EN-CR so assigning this as the overall status for Criterion D was consistent. However in this study, in many cases the risk level for corals was LC while that for groupers was EN-CR, and we could not justify assigning a risk category of EN-CR to an entire ecoregion's reefs on this basis.

Based on our ecosystem model and the compartments used (fig. 2a), we propose an algorithm of sequential steps for assessing criterion D, and based on the stepwise increases in risk levels VU-NT-VU-EN-CR (see fig. 1). Our algorithm establishes coral cover as the 'root variable' for setting the base state of a coral reef ecosystem, following which 'strong interactions' are considered in sequence – first competition with algae, then top-down control of algae by herbivores (parrotfish) and finally apex predator interactions by piscivores (groupers). For each step in this sequence, the initial risk status may be raised based on the following logic:

1. If the risk status of the next compartment is the same as, or less than, that of the prior compartment (if coral) or aggregate (if algae or parrotfish have been considered), the current risk status is conserved;
2. If the risk status of the next compartment is higher than that of the prior compartment (if coral) or aggregate (if algae or parrotfish have been considered), the current risk status is increased by one step, irrespective of the gap in status between the two.

Thus, given the model and data available here, the coral risk status sets the basis, then first algae:coral ratio, then parrotfish then grouper status might increase the aggregate level of risk by a single category at each step (Table S2).

Table S2. Multi-compartment algorithm for assessing Criterion D. For full justification, see the main text in "Coral reef ecosystem model" and Table S16 for worked examples.

Compartment	Explanation	Role	Algorithm
Coral (all hard corals, % cover)	Is the primary indicator of coral reef state as it describes the dominant ecosystem engineers, and is the most commonly and accurately measured indicator [41]. If coral composition information were available, compositional change away from fast growing, habitat-forming species could result in a higher initial risk level for corals.	Ecosystem engineer	Coral cover gives the first estimate of risk of collapse for Criterion D. If available, coral composition changes may raise the risk level based just on total cover.
Fleshy algae (turf, macro- and calcareous algae combined, % cover)	The health of corals, and their functional role in coral reefs is primarily challenged by fleshy algae. However, algae is measured more variably, and data are less available.	Second step, modulator of status given by coral cover	If at a higher risk of collapse than corals, it raises the risk of collapse one category higher.
Parrotfish abundance	Herbivores are a dominant mediator of coral-algae interactions, in parallel with invertebrates such as sea urchins. How and which fish	Third step, modulator of	If at a higher risk of collapse than corals + algae, raises the

(herbivory indicator)	populations are measured is highly variable. Abundance is the most common metric in the WIO.	algae-coral interactions	risk of collapse one category higher.
Grouper abundance (piscivory indicator)	Piscivores are strong controllers of reef communities, mediating energy transfer and biomass accumulation with cascading impacts on the benthos. How and which fish populations are measured is highly variable. Abundance is the most common metric in the WIO.	Fourth step, modulator of complex interactions	If at a higher risk of collapse than corals + algae + parrotfish, raises the risk of collapse one category higher.

Specific collapse thresholds for each criterion are presented in the criterion sections below, and the thresholds are listed in Table 1.

We tested this 'sequential' algorithm against two others, described in section 6.1.4.

2.7. Whole-region assessment

For criteria A, C and D we calculated a result for the Western Indian Ocean region as a whole, in addition to for the ecoregions individually. We tested three methods, to assess the impact of the unequal representation of sites and coral reef area among ecoregions (see Tables S3, S6, S12-14). This allowed us to explore the differences in the results and make a more robust interpretation. The three methods were:

1. As one region – treat all sites as a representative sample across the whole region and undertake the full iterative analyses.
2. Unweighted average of ecoregions – calculate the average across ecoregional results with equal weighting.
3. Weighted average of ecoregions – weight the result from each ecoregion by its proportional area of reef in the WIO (Table S1).

For criterion C, because of the nature of the data, only methods 2 and 3 were used, from which the unweighted and weighted relative severity for the region was calculated and assumed to occur over 100% of the region.

Given the highly unequal area of reef among the ecoregions, we determined that the weighted result (method 3) is most representative, so is presented as the final result. This has the effect of increasing the weighting of the large-reef-area ecoregions on the mainland coast and reducing the weighting of the small-reef-area islands and peripheral ecoregions.

2.8. Data limitations and uncertainty

The RLE guidelines [1] give specific guidance on standards of evidence for dealing with uncertainty (section 3.3.3) and using expert knowledge as well as quantitative data (section 3.3.4). We address these in relation to the data and model parametrization for each Criterion in the relevant sections below, and present an overview in section 7.1.

3. Criterion A – Reduction in geographic distribution

3.1. Methods

Criterion A is based on the area or extent of an ecosystem, and its change over 50 years. When it comes to assessing the change in area, coral reefs present unique challenges compared to other ecosystems such as forests. A coral reef consists of two features – the geo-morphological rocky reef (which includes its three-dimensional structure and areal extent), and the dominance and functional role of hard corals (which contribute to building this physical structure, and in providing space for diverse biotic interactions). The areal extent of coral reefs can be obtained from satellite-derived products, such as the Millennium coral reef maps [9], but the rigid reef structure generally only changes due to local developments (port development, land reclamation) or cyclones. Additionally, the percent cover of coral on a reef surface cannot be estimated from current satellite remote sensing, meaning that a reef can functionally transform without any detected change in structure (or extent) by mapping. Further, there is no consistent and geographically complete time series of either facet of coral reefs to assess reduction in geographic distribution over the last 50 years.

We could only evaluate Criterion A1, of coral reef decline over the last 50 years. Criteria A2a, A2b or A3, requiring assessment into the future, could not be assessed. We assume that all sites defined as 'coral reefs' and included in monitoring programmes were functioning coral reefs and thus above the threshold 50 years ago. On this basis, we excluded just one station which had coral cover below 10% before 1998. Current estimates of coral cover values were obtained by averaging across a window of 7 years (i.e. 2013-2019), because monitoring has occurred inconsistently across the region. This wider temporal period allowed us to a) increase the number of sites assessed (to include ≈ 65% of 879 sites) and b) even out the influence of inter-annual inconsistencies and changes in coral cover at sites (e.g. due to major coral bleaching in 2016). If the current value was < 10%, the site was classified as collapsed. For each eco-region, the percentage of collapsed sites compared to the total sites was assessed against the thresholds for Criterion A (Table 1). To explore the influence of the threshold coral cover for collapse, we repeated the analysis for thresholds from 10% down to 1% coral cover in unit reductions.

3.2. Intermediate results

The threat status for the entire WIO was assessed using three approaches, with all of them showing LC status (range 12-18 %, Table S3). Given the differences in reefs across the ecoregions, different area of reef in each ecoregion and different levels of sampling among them, we consider the weighted mean to be a more accurate representation of the status of the entire region, as the LC condition of the ecoregions with high reef area should outweigh the more threatened status of ecoregions with less reef area.

Table S3. Proportion of coral reef sites collapsed at a threshold of 10% coral cover for each ecoregion, and the Western Indian Ocean as a whole.

Ecoregion		Loss %	Category
1	N.Tanzania-Kenya	17	LC
2	N.Mozambique-S.Tanzania	4	LC
3	Comoros	4	LC
4	West Madagascar	13	LC
5	North Madagascar	0	LC
6	Seychelles Outer	30	VU
7	Seychelles North	31	VU
8	Mascarene Islands	4	LC
9	East Madagascar	0	LC
10	South Madagascar		DD

11	Delagoa	40	VU
Western Indian Ocean (Province)			
Weighted average of ecoregions		12	LC
Unweighted average of ecoregions		14	LC
As one region		18	LC

Seven ecoregions were Least Concern (LC) (Table S3): five experienced < 5% and two experienced 15% reduction in area (North Tanzania-Kenya and West Madagascar). Three ecoregions were Vulnerable (VU) (Seychelles Outer (30% reduction), Seychelles North (31%) and Delagoa (40%)). South Madagascar was Data Deficient (DD) for coral cover.

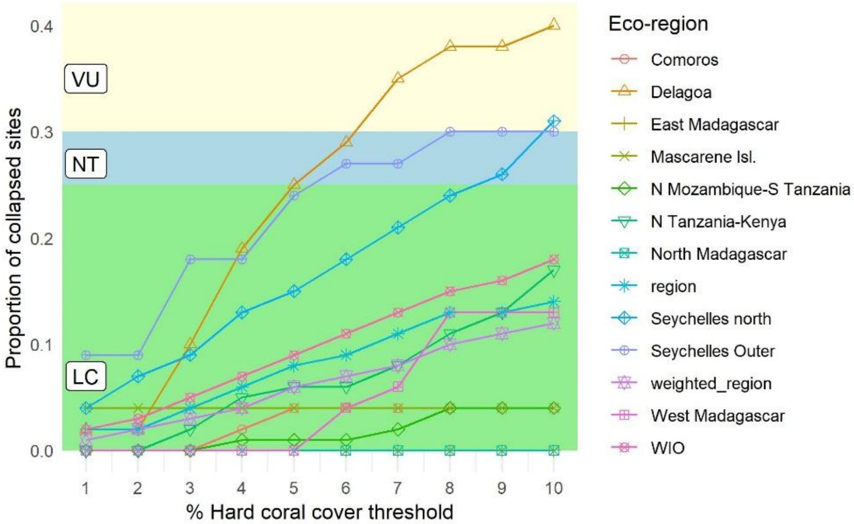

Figure S3. Sensitivity analysis of Criterion A assessment to the collapse threshold for coral cover varying from 1% to 10%.

Sensitivity of the analysis to the threshold coral cover below which a coral reef can be said to be collapsed is moderate (fig. S3). Delagoa crosses the threshold to VU at 7% coral cover, Seychelles Outer at 8%, and Seychelles North at 10%. If a threshold of 5% coral cover is used, then 2 ecoregions (Delagoa, Seychelles Outer) may be considered NT, while Seychelles North would be considered LC. Nevertheless, all three ecoregions showed steeper increases in the proportion of reef collapsed, compared to the other ecoregions from 3% coral cover and above. On this basis we consider the greater vulnerability of the 3 VU ecoregions to be a robust result.

3.3. Methodological discussion

The conventional method for assessing Criterion A, although straightforward and simple to apply for most terrestrial ecosystems, presents challenges for marine ecosystems due to the difficulties with remote-sensed mapping of underwater features. An extensive global coral reef map exists. However, it has been mapped at broad scales and with minimal ground-truthing so the areal extent estimate is not reliable, and there is no time-series to make comparisons over the required 50-year time-period.

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Inherent biases with recent monitoring of coral reefs exist which affect the method applied: due to a tendency for new programmes to select sites which are in relatively good health, and/or to discontinue monitoring at sites which have degraded and showed no signs of recovery, this approach may underestimate the true distribution of sites which have declined below the threshold value since before major impacts in the 1980s/1990s. This approach would be improved with either random or uniform distribution of sampling sites, and inclusion of sites which have degraded historically.

Some sites have naturally low coral cover, such as Kiunga in Northern Kenya, due to high productivity which promotes algal dominance; 3 forereef sites surveyed in the 1980s had a mean coral cover of 3.4 % [45], and persisted in this state through the early 2000s [46]. In Delagoa, soft coral or algae dominate reefs, as the region is marginal for coral reefs [47], so hard coral cover is generally low. However a real decline in coral cover crossing the threshold of 10% was documented in Ponta d'Ouro in the southern part of Mozambique, where tourism and coastal development pressure are high, and reef degradation has been documented [48].

4. Criterion B – Restricted geographic distribution

4.1. Methods

A restricted geographic distribution, or limited area, is a key determinant of ecosystem vulnerability, as any major threat may affect a large proportion of the overall ecosystem extent. The Millennium coral reef layer [9] was used to define the spatial extent of coral reefs, with some modifications as listed under section "Area of assessment" (section 2.1). The RLE uses two standard measures for area (Table S1):

- Extent of Occurrence (EOO), the minimum convex polygon around all the points at which the ecosystem is found (Criterion B1).
- Area of Occurrence (AOO), the count of 10*10 km grid cells that contain a minimum of 1% of their area in the coral reef ecosystem layer. The fit of the reef layer to a grid varies with the positioning of the grid; to avoid overestimation of AOO and misclassification of risk category, we derived the minimum grid uncertainty values by repositioning the 10x10 grid four times and using the minimum (Criterion B2).
- Criterion B3, based on the number of threat-defined locations was not assessed. The two principal threats indicated by the literature, climate change (warming) and fishing, are assessed quantitatively in criteria C and D, respectively. Fishing occurs through localized actions, aggregating these to estimate a large scale threat in relation to coral reef area is not easily done. In relation to thermal stress the size of hot-spots of high water temperature varies greatly from year to year and in the hottest years may extend across ecoregions. However given the high variability and influence of localized factors in influencing thermal stress, both globally [49] and documented in the region [50] we judged that Criterion C gives a less subjective assessment of this threat than B3, thus more appropriate for analysis.

AOO and EOO for each ecoregion were obtained (Table S1) and assessed against the standard thresholds established for the RLE (Table 1) [1]. Criterion B1 and B2 require one of three sub-criteria to be met, and the following two were found to be relevant for this assessment:

a(iii) "*a measure of disruption to biotic interactions appropriate to the characteristic biota of the ecosystem*" is documented by the analysis for Criterion D through coral cover decline, and also through the literature (see sections 2.2-2.4).

b "*observed or inferred threatening processes that are likely to cause continuing declines in geographic distribution, environmental quality or biotic interactions within the next 20 years*" being justified by observations of historic coral bleaching and rising sea surface temperatures (see sections 2.2-2.4) as well as projections of continuing declines in the near future as documented under Criterion C.

4.2. Intermediate results

Coral reefs were assessed at higher risk for B2 than B1 (Table S4), with seven ecoregions assessed as LC, three VU and one EN for B2, and nine LC and two VU for B1. Taking the most conservative of these results for an overall analysis for Criterion B, seven were LC, three were VU (Mascarene Islands, East Madagascar, Delagoa) and one EN (South Madagascar).

Table S4. Assessment of Extent of Occurrence (EOO) and Area of Occurrence (AOO) for coral reef ecoregions in the Western Indian Ocean. See Table 1 for threshold levels for criteria B1 and B2.

Ecoregion		**EOO**		**AOO**		Overall
		(km^2)	**Status**	**# grid cells**	**Status**	
1	N.Tanzania-Kenya	76,699	LC	198	LC	LC
2	N.Mozambique-S.Tanzania	174,710	LC	270	LC	LC
3	Comoros	57,081	LC	84	LC	LC
4	West Madagascar	410,367	LC	184	LC	LC
5	North Madagascar	64,829	LC	166	LC	LC
6	Seychelles Outer	683,231	LC	80	LC	LC
7	Seychelles North	130,878	LC	79	LC	LC
8	Mascarene Islands	73,785	LC	45	VU	VU
9	East Madagascar	34,464	VU	96	LC	VU
10	South Madagascar	39,585	VU	12	EN	EN
11	Delagoa	73,934	LC	42	VU	VU

4.3. Methodological discussion

The results are sensitive to scale because below a minimum ecoregion size and potential for reef area, criteria B1 and B2 are automatically triggered, and above a certain size makes the criteria irrelevant. The smallest ecoregion by area of coral reef (Table S1), South Madagascar, was also the most threatened under this Criterion, although the second smallest, Delagoa, has a more spatially spread reef system, and was not as threatened. This analysis used the same ecoregion definition for East Madagascar as a national RLE assessment in Madagascar [7], but because of the addition through a review process of one small isolated reef, far south of other reefs in the ecoregion, our EOO has considerably larger, triggering VU where their analysis triggered an EN classification. There were 2 more eco-regions which came out as threatened using AOO compared to EOO.

Spatial data gaps (un-mapped reefs) in the Millennium coral reef layer, may be driving the vulnerable status in some of the southern reefs e.g. south of Inhambane, Mozambique (Inhaca-Inhambane area) in the Delagoa ecoregion. Additionally, the classifications in the coral reef layer also includes areas which may not be true coral reefs or coral dominated, therefore overestimating the presence/distribution of reefs, and potentially underestimating the threat status. There is an opportunity to re-visit the analysis using newly produced and more accurate global coral reef layers derived from advanced satellite and processing technology and techniques (e.g. the Allen Coral Atlas, https://allencoralatlas.org/)

5. Criterion C – Environmental degradation

5.1. Methods

Abiotic degradation reduces the capacity of an ecosystem to sustain its characteristic biota. Applying our ecosystem model (fig. 2a), increasing seawater temperatures reduce the environmental suitability for reef-building corals. SST warming has produced mass coral bleaching and mortality events within the WIO [14,21] and globally [51,52] and is well established as the dominant threat to coral reefs into the future [10,53]. The roles of sedimentation and eutrophication (using Chlorophyll-*a* as a proxy) in reducing suitability of conditions for coral reefs at local-scales were also considered, but data series were not of sufficient temporal resolution (i.e. 50 years), nor with clear thresholds for collapse to allow their use.

Historical temperature data are not available over a period of 50 years so C1 (50 years) and C3 (from 1750 to today) could not be evaluated. However, projected sea surface temperature data, already interpreted for risk of large scale coral mortality and ecosystem collapse, is available for > 50-year duration [54], enabling its use for Criterion C2a. The projected and historical temperature datasets are very different in nature so they cannot be evaluated together as required for C2b. Thus, only C2a was evaluated.

We used an indicator of thermal stress derived for corals, Degree-Heating-Weeks (DHW) [55]. The relationship between Degree Heating Weeks and coral bleaching and mortality is well established, including in projecting future conditions for coral reefs [53,56]. DHW is computed as the excess in night-time temperature over 1°C above the climatological Maximum Monthly Mean for a location, accumulated over a moving 12-week window. The model used SST projections from 2010 to 2100, from climate model outputs adjusted to the mean and annual cycle of observations of SST based on the OISST V2 1982-2005 climatology (as in [57]). Degree heating months were calculated by summing the positive anomalies above the warmest monthly temperature from the OISST V2 1982-2005 climatology [58] for each 3-month period. Degree heating months are then converted into DHWs by multiplying by 4.35 (see also [57,59]. DHWs were re-gridded to the same grid as the coral reef layer, using cdo remapbil [60]. We extracted the maximum DHW for each coral reef-containing pixel for each year from 2010 to 2100, and calculated the average for each ecoregion and year.

To cover a 50-year time period in the future, we assessed 2070 against 2020. A collapse threshold of 2 major bleaching events per decade (i.e. two exceedances of the DHW threshold) was used as a limit for ecosystem collapse [44](Table 1). This is equivalent to one major bleaching event every 5 years, which if sustained would limit recovery and lead to continued degradation and collapse. We counted the number of exceedances of the DHW thresholds for the decades spanning the current (initial) year, 2020 (2015-2024) and fifty years from now, 2070 (2065-2074), to calculate relative severity.

For each ecoregion we calculated the relative severity as:

Relative severity (%) = (predicted decline / Maximum decline) × 100

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where

predicted decline = Initial (2020) DHW value –future (2070) DHW value &

Maximum decline = Initial (2020) DHW value – Collapse value

and, Collapse value = 2.

The DHW values used were an average across the entire ecoregion's coral reef area, so we assume that the relative severity was calculated over 100% of the ecoregion. Therefore, the threat categories defined by the RLE Guidelines are those for > 80% extent [1](Table 1). We evaluated the threat status for each eco-region for both levels of heat stress (DHW 8 and 12) and for each of the four climate change scenarios (Table S5).

Table S5. Scenarios for carbon dioxide emissions (and therefore warming) and heat stress thresholds used in evaluating Criterion C.

Heat stress	***Scenarios of climate change –***
As heat stress increases, bleaching becomes more severe, passing a threshold at which moderate mortality is likely, then severe mortality (Liu 2012). ● Moderate mortality is likely from approximately DHW = 8 and higher, but some bleached corals do recover. ● Severe mortality is likely from approximately DHW = 12 and higher, with many examples of catastrophic mortality of corals above this, with very little recovery of bleached corals.	Representative Concentration Pathways (RCPs) are scenarios for CO_2 emissions established by the IPCC, presenting possible futures to illustrate potential impacts of climate change. ● RCP 2.6 is a 'best case' scenario of strong CO2 mitigation, and necessary to achieve the Paris Agreement; ● RCPs 4.5 and 6.0 represent different intermediate cases with an earlier peak emission and lower level of long-term accumulation in RCP 4.5 (peak emissions in 2040s). Peak emissions in RCP 6.0 occur around 2080. Projected SST to about 2050 is very similar between them. ● RCP 8.5 is a 'business as usual' scenario of minimum carbon emission reduction – with high accumulation of CO2 in the atmosphere

5.2. Intermediate results

Results were explored for both DHW 8 and 12, and all four RCP scenarios (Table S6). At a critical threshold of DHW8, all ecoregions were LC for RCP 2.6, but all turn CR for the moderate climate change scenario RCP 4.5 and above. For the critical threshold of DHW 12, all ecoregions were less threatened than at DHW 8. For RCP 2.6 all ecoregions remained LC. At RCP 4.5 two ecoregions shifted to EN and four became CR. At RCP 6.0 three ecoregions were EN and four CR. Interestingly Seychelles Outer was less threatened (EN) under RCP6.0 than RCP 4.5 (CR). At RCP 8.5 all ecoregions were CR.

Table S6. Risk levels for each ecoregion based on four climate scenarios and two thresholds of collapse from thermal stress.

		DHW 8				**DHW 12**			
	Ecoregion	**RCP2.6**	**RCP4.5**	**RCP6.0**	**RCP8.5**	**RCP2.6**	**RCP4.5**	**RCP6.0**	**RCP8.5**
1	N.Tanzania-Kenya	LC	CR	CR	CR	LC	LC	LC	CR
2	N.Mozambique-S.Tanzania	LC	CR	CR	CR	LC	LC	LC	CR
3	Comoros	LC	CR	CR	CR	LC	CR	CR	CR
4	West Madagascar	LC	CR	CR	CR	LC	EN	EN	CR
5	North Madagascar	LC	CR	CR	CR	LC	EN	EN	CR
6	Seychelles Outer	LC	CR	CR	CR	LC	CR	EN	CR

7	Seychelles North	LC	CR	CR	CR	LC	LC	LC	CR
8	Mascarene Islands	LC	CR	CR	CR	LC	LC	CR	CR
9	East Madagascar	LC	CR	CR	CR	LC	CR	CR	CR
10	South Madagascar	LC	CR	CR	CR	LC	CR	CR	CR
11	Delagoa	LC	CR	CR	CR	LC	LC	LC	CR
WIO region									
Unweighted average of ecoregion		LC	CR	CR	CR	LC	VU	EN	CR
Weighted average of ecoregions		LC	CR	CR	CR	LC	VU	VU	CR

We elected to use the results from DHW 12 as this represents a more conservative threshold for reef decline. At DHW 8 while stress to coral colonies is significant and mortality occurs, the chance for recovery is higher. Field studies have found significant threshold effects for coral reef state at lower levels of thermal stress, such as DHW 4 [61] and DHW 6[62], but more conservative and severe thresholds for projected data are advised[54,63], so we used the DHW 8 and 12 thresholds advised by Liu et al.[55] . Also, given acclimation and adaptation to rising temperatures that has been observed in the last 20 years (e.g. [64–66]) it may be that DHW 8 may not represent a significant stress in 50 years time. DHW12 is associated with more severe impacts so represents a more conservative threshold to use for long term change.

We elected to use the results from the intermediate climate change scenario RCP 6.0. Current evidence indicates RCP 6.0 is a closer fit to current emissions than higher or lower emission scenarios 8.5 [67–71], and the observed CO_2 concentration in 2020 was very close to that projected for RCP6.0, of 409 ppm. Comparing calculations of risk for each ecoregion for RCPs 4.5 and 6.0, the latter more closely matched evidence of climate vulnerability (Table S6). The Mascarene Islands were LC for RCP 4.5 and worsening abruptly to CR for RCP 6.0, while the Seychelles Outer was CR for RCP 4.5 improving to EN for RCP 6.0. Given observations to date of coral bleaching and mortality in the Mascarene Islands [72] we find the rating of LC for RCP 4.5 to be unrealistic, and the between-ecoregion comparisons for RCP 6.0 more realistic. Finally, the results for RCP 2.6 (all ecoregions LC) and 8.5 (all ecoregions CR) give no opportunity for comparisons among ecoregions, greater understanding of their sensitivities, and interpretation of the results to derive options for responses.

Accordingly, we selected DHW 12 and RCP 6.0 results for further analysis.

For the whole-region assessment of Criterion C we applied methods 2 and 3, as the results were not available in a format to allow method 1. For DHW 8, both unweighted and weighted approaches gave a result of LC for RCP 2.6 and CR for the other three scenarios. For DHW 12, both approaches gave a result of LC for RCP 2.6, VU for RCP 4.5 and CR for RCP 8.5. The weighted approach gave a result of VU for RCP 6.0, reflecting the greater weight to the less threatened larger reef-area ecoregions, while the unweighted average gave a result of EN for RCP 6.5.

5.3. Methodological discussion

Time series for observed SST data and projections cannot be combined or directly compared [54,63,73], hence our inability to use Criterion C2b. A key point of difference is that the projected datasets show much lower amplitude of variations, and that these are smoothed over large grid cells in the Global Climate Models, so application of the same criteria for calculated thermal stress through Degree Heating Weeks [55] incorporates many assumptions. For example, Seychelles North is the part of Eastern Africa most strongly impacted by high thermal stress and mass mortality of corals to date [14,15,74] yet this analysis assesses it as LC, comparable to N.Tanzania-Kenya and N.Mozambique-S.Tanzania, which are established in other analyses to have comparatively good climate futures [10], and Delagoa, which has lower temperatures and parts of which have experienced less thermal stress than other parts of the region [75,76]. This may indicate limitations with broad-scale, global climate models in picking up localized effects such as heating on the Seychelles banks, or the effect of the ecoregion size which includes a large oceanic area much larger than just the banks and inner islands, potentially masking heating affecting coral reefs. New and improved models and data can be used to update the analysis and produce more accurate results.

Further caveats to our application of thermal stress thresholds applicable today are that acclimation and adaptation by corals and zooxanthellae to warming temperatures are not considered, though they have been observed in the region [66] and elsewhere [64,65]. While the 50-year timespan of this analysis may provide significant opportunity for further adaptation, the degree of trait shift needed for corals to cope with the degree of warming expected is likely to be too high [77,78].

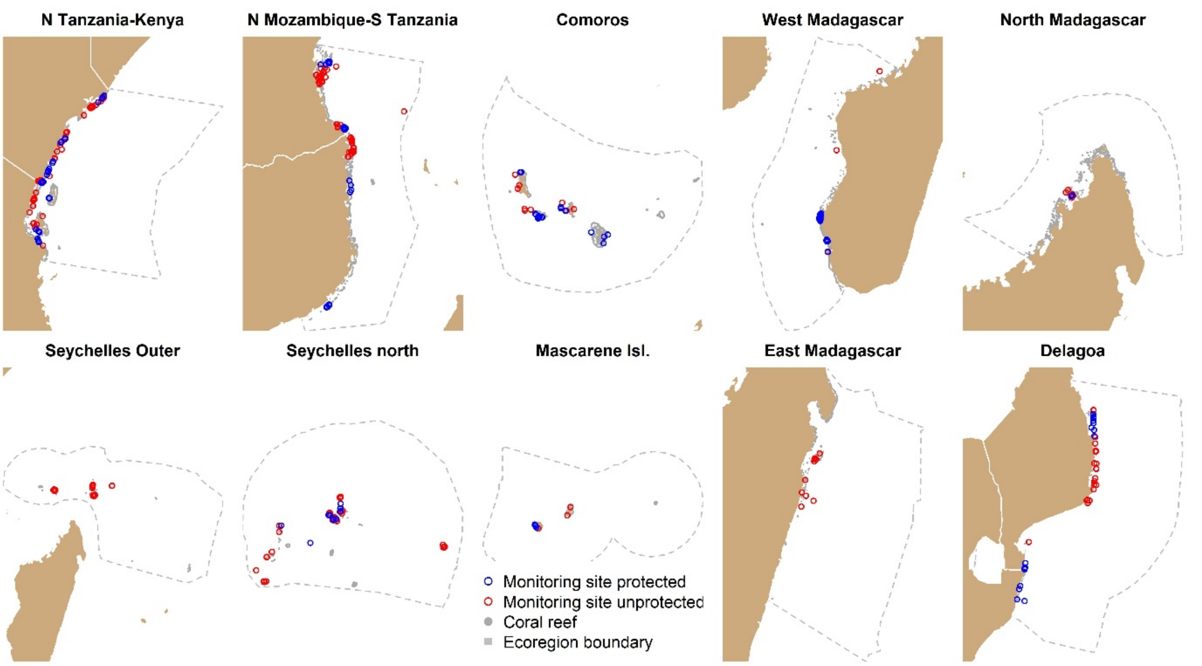

Figure S4. Distribution of benthic monitoring sites across coral reefs (grey) in the 10 eco-regions with data. Blue circles are sites within protected areas and red circles are unprotected sites. Algae and fish data were obtained from smaller subsets of these sites.

6. Criterion D – Biotic disruption

6.1. Methods

Disruption to biotic processes and interactions leads to loss of function in an ecosystem and its potential collapse. This is particularly important for processes and/or organisms playing key functional roles. Applying our ecosystem model (fig. 2a), we assess four main compartments of the coral reef: i) hard corals as the engineers of coral reef ecosystems ii) fleshy algae, as the principal competitor to coral, iii) parrotfish which provide strong mediating effects on algae and corals, and iv) groupers which as piscivores impose functional and trophic controls of multiple aspects of reef ecology. Reef rugosity is an important biophysical characteristic determining reef processes, partly incorporated in Criterion D as it is partially dependent on live coral growth [42], but there are no consistent data collected on rugosity at regional scales.

Monitoring of coral reefs has focused on measurements of hard coral cover (and composition), algal cover and fish abundance and biomass for many years, and these variables have been recognised as standard measures of ecosystem functioning [41]. The Global Coral Reef Monitoring Network (GCRMN) is the main aggregator of coral reef data within this region [15,21] and globally [41,79]. Together with additional contributed data, the GCRMN dataset for the WIO provided the four variables used in the Criterion D assessment. Fig. S4 shows the spread of sites at which coral data was obtained across the ten ecoregions, with sample sizes shown in respective tables below. Monitoring sites were principally located in sheltered lagoon reefs or exposed fringing reefs, which have been lumped together in regional analyses due to common patterns in benthic cover and incomplete sampling if reef zones are separated in analyses (see section 2.1 and [15,21]).

<u>Hard coral cover</u> – the percent cover of hard corals, the principal measure for coral reef health going back several decades [41,79].

<u>*Coral – algae ratio*</u> – the percent cover of fleshy algae (which may include turf, macro and calcareous algae) represents, at the highest level, the main competitive cover category to coral cover, with strong interactions between the two. In the regional dataset we used, turf, calcareous and macro-algae were combined into a single group 'harmful to corals', due to multiple sources of variation among contributed datasets [15,80]: some datasets only reported the broader group 'algae'; differences in distinguishing turf from 'macro' algae; differences in terminology used (e.g. 'turf' vs 'epilithic algal community', 'macro vs 'fleshy' algae', 'turf' vs 'bare substrate' vs 'rock'. Interactions between fleshy algae and high-canopy turfs, and corals, are multiple [81,82], including– overgrowth, abrasion and competition by algae with larger fronds (macro, calcareous), as well as by thick turfs at the algae-coral border; leaching of compounds from turf and fleshy algae may impede coral growth by stimulating microbial growth [83]; direct contact and transfer of disease-causing or stress-inducing agents from the algal community (which includes diverse microbial community and many small invertebrates); and inhibition of coral recruitment [84]. While thin algal turfs (< 2 mm) are not competitive to corals, active grazing by herbivores (e.g. sea urchins, and some grazing/excavating fish) may damage adjacent coral tissues, particularly at high levels documented on many East Africa reefs [85]. As a result, at the broader taxonomic and functional level of this reef model, given the data limitations in the regional dataset, and that identification of 'turf' algae tends to be more inclusive of thicker rather than tinner turfs, we applied practice in Obura et al. 2017 [15] to combine turf, calcareous and macro-algae into a single category, 'fleshy algae'. Improvements in datasets to distinguish high from low-canopy turfs [81], and the full suite of algal functional groups, and narrowing analysis down to smaller scales resulting in greater consistency among contributing datasets, will enable differentiation of algal functional groups within the broader algae compartment (fig. 2).

Because of the interplay between corals and algae, it is not just the cover of algae that is critical, but the degree to which algae cover exceeds coral cover. Thus, we adopted the algae:coral ratio (ACR) [86], obtained as ACR = Algae cover/ (Algae + Coral cover). This index is useful as it is bound between zero and 1, when coral cover is higher it is necessarily low, and when coral cover is low it can become high if algae dominates the substrate.

Fish abundance – the number of individuals of functional or taxonomic groups of fish within a standard area (see Methods: Criterion D – Biotic disruption), for herbivorous and piscivorous fish. With the data available we identified one family to represent each of these groups - parrotfish (tribe: Scarini within family Labridae) and groupers (family: Epinephelidae), respectively.

Parrotfish are nominally larger bodied herbivores, with a mean trophic value of 2.00 in the WIO [6]. Using Bellwood and co-authors' [87] functional definitions, all parrotfish species (except *Calotomus* spp. and *Leptoscarus*) are Grazers which remove small turf algae, sediment and bio-erode the reef by scraping or excavating the reef matrix thereby transferring energy and material in large quantities [88]. However, other taxa, particularly Acanthuridae and Siganidae are also Grazers, though not all species in these two families are. Due to the constraints of only family level identification of fishes in much of the WIO GCRMN data, the Scarini or parrotfishes [89] were selected because they were well represented in most data sets. Scarini comprise 49.5% of all herbivorous reef fishes based on species level data from the WIO [6] and herbivore categories (turf and macro algae removers) [87].

Groupers represent relatively large bodied, high level consumers [90,91] which exert top down control on lower trophic level taxa, can occur in significant numbers in reef environments and also move en masse which involves considerable energy transfer (1,2)[35,92]. Groupers represent one of the highest level trophic groups for reef associated fishes, hence a significant energy store [93] with a mean trophic level of 4.23 recorded in the WIO [6]. Grouper regulate prey communities, are closely linked to reef structure and their loss has been shown to affect reef ecosystems directly and indirectly [35,94,95]. Groupers are also vulnerable to fishing which is one of the key threats on coral reefs globally [96,97]. The proportion of piscivore individuals that epinephelids represent in the WIO is 82.3% [6] based on 12 of the 15 families of reef associated fish families in the GCRMN data [15]. Piscivores have previously been used for Criterion D in a RLE assessment of coral reef ecosystem health for the Meso-American Reef and found to be the most sensitive of two fish indicators [44].

Coral and algae data were available for 10 of the 11 ecoregions, while fish abundance data were available for 7 (groupers) and 6 (parrotfish) ecoregions. South Madagascar was data deficient across all 4 variables for Criterion D.

6.1.1. Current and Initial values

We assessed change in biotic variables over the past 50 years, thus Criterion D1. Criteria D2a and D2b were not assessed as future values for the above variables cannot be projected reliably, and D3 was not assessed as data are not available as far back as 1750.

Monitoring across sites in the WIO has increased but data are sparse before 1998. There are some datasets from a small number of sites (Kenya, Zanzibar, South Africa) starting in the late

1980s/early 1990s. The most recent year of data collection varied across sites so a window spanning the last 7 years (2013-2019) was used to calculate a mean for each site, to represent current conditions. This also has the effect of smoothing inter-annual inconsistencies and short-term fluctuations in benthic cover and fish abundance (e.g. due to bleaching in 2016). The mean, standard deviation and maxima/minima of current values are shown for each ecoregion for illustration (Tables S7-9), but it is the actual site-level values that were used to assess risk of collapse.

Because monitoring data were not available from 50 years ago (pre-1970), we extrapolated back in time to estimate initial conditions. For coral and algae cover, we used data from sites known to be in healthy condition prior to (or just after) the 1998 mass bleaching event to estimate initial conditions for each ecoregion (Tables S7, S8). For these sites we assumed that there was only minor changes in reef benthic composition before the 1998 bleaching event thus their condition was equivalent to conditions around the baseline year of 1970. West Madagascar ecoregion had no initial data for hard coral cover, so this was estimated from the mean for the other 3 Madagascar ecoregions. Some of the ecoregions only had a single data point and so a common standard deviation was calculated for all ecoregions by averaging across standard deviations for all ecoregions (sd= 13.5%). For algae:coral-ratio, there was even less initial data for algae cover, so we calculated a single initial ratio for each ecoregion (0.2754, sd= 0.151, n=17 sites) except for Kenya-N. Tanzania, which had enough data (29 sites) to calculate an independent value (0.431±0.191). For fish abundance, data were analysed for those ecoregions that contained well protected sites, since these were required for calculation of initial population values (see below).

Table S7. Initial and current values of hard coral cover for each ecoregion. Reference values were based on published and in-house data from healthy reef sites before 1998. Current values were based on data from 2013-2019, and the number of sites 'n' is shown.

	Initial coral cover, %				**Current coral cover, %**					
Ecoregion	**Mean**	**SD**	**cv**	**n**	**Mean**	**SD**	**cv**	**Max**	**Min**	**n**
N.Tanzania-Kenya	37.2	14.1	0.4	49	29.8	18.5	0.6	86.0	2.3	113
N.Mozambique-S.Tanzania	44.2	16.0	0.4	29	36.8	17.2	0.5	80.4	4.0	98
Comoros	56.5	4.4	0.1	4	41.0	19.4	0.5	79.0	3.3	48
West Madagascar	50.9		n/a	*0	25.0	13.7	0.5	56.8	5.3	47
North Madagascar	50.9		n/a	1	42.4	13.0	0.3	58.3	17.6	13
Seychelles Outer	47.0	17.5	0.4	3	24.2	17.6	0.7	64.0	1.0	33
Seychelles North	30.3	10.6	0.3	10	21.2	15.9	0.8	84.0	0.0	137
Mascarene Islands	43.2	9.1	0.2	21	40.8	13.6	0.3	60.0	0.3	23
East Madagascar	47.1		n/a	1	30.2	12.3	0.4	50.5	13.0	14
South Madagascar	54.7		n/a	1	n/a	n/a	n/a	n/a	n/a	0
Delagoa	39.6	22.6	0.6	6	16.1	12.2	0.8	48.8	0.0	48

Table S8. Initial and current values of algae:coral ratio for each ecoregion. Reference values were based on published and in-house data from healthy reef sites before 1998. Current values were based on data from 2013-2019, and the number of sites is shown.

Ecoregion	**Initial algae:coral ratio**			**Current algae:coral ratio**					
	mean	SD	cv	mean	SD	cv	Max	Min	n

N.Tanzania-Kenya	0.43	0.19	0.4	0.50	0.30	0.6	0.96	0.00	112
N.Mozambique-S.Tanzania	0.28	0.15	0.5	0.31	0.22	0.7	0.95	0.00	93
Comoros	0.28	0.15	0.5	0.50	0.25	0.5	0.95	0.02	36
West Madagascar	0.28	0.15	0.5	0.62	0.22	0.4	0.92	0.00	47
North Madagascar	0.28	0.15	0.5	0.12	0.07	0.6	0.22	0.02	13
Seychelles Outer	0.28	0.15	0.5	0.18	0.32	1.8	0.93	0.00	33
Seychelles North	0.28	0.15	0.5	0.35	0.32	0.9	1.00	0.00	116
Mascarene Islands	0.28	0.15	0.5	0.45	0.23	0.5	0.99	0.01	22
East Madagascar	0.28	0.15	0.5	0.27	0.25	0.9	0.65	0.00	14
South Madagascar	0.28	0.15	0.5	n/a	n/a	n/a	n/a	n/a	0
Delagoa	0.28	0.15	0.5	0.61	0.29	0.5	1.00	0.00	42

Table S9. Current values of fish abundance (indiv./ha) by ecoregion, for Criterion D. Presented are mean, standard deviations (SD) and coefficient of variation (cv), and collapse threshold values for parrotfish and grouper. The collapse thresholds are 10% (parrotfish) and 20% (grouper) of initial value (see main text).

Fish family	**Eco region**	**No. of sites**	**Current abundance (indiv/ha)**			**Collapse threshold**
			Mean	SD	cv	
Groupers	N.Tanzania-Kenya	30	60.4	53.3	0.9	35.7
	N.Mozambique-S.Tanzania	9	56.0	38.6	0.7	50.6
	Comoros	37	373.2	989.0	2.7	75.8
	West Madagascar	24	24.1	13.3	0.6	39.5
	Seychelles North	28	122.8	46.4	0.4	33.6
	Mascarene Islands	14	34.9	21.7	0.6	15.4
	Delagoa	32	9642.8	8785.3	0.9	278.0
Parrotfish	N.Tanzania-Kenya	44	845.6	1163.5	1.4	62.9
	N.Mozambique-S.Tanzania	12	835.3	488.0	0.6	86.7
	Comoros	35	379.1	291.3	0.8	49.3
	West Madagascar	33	123.5	71.0	0.6	25.6
	Seychelles North	43	1090.9	575.6	0.5	91.5
	Delagoa	27	221.6	174.0	0.8	50.2

Initial values for fish abundance were derived using two sets of values that approximated healthy baseline fish populations. First, across all ecoregions, we used two remote, protected and uninhabited reef areas as reference sites: Chagos Archipelago, central Indian Ocean [98] and Iles Glorieuses, WIO [99] (Table S10). These we considered to represent close to intact fish populations with respect to fishing, and have also been formally protected from fishing since 2010 and 2012, respectively. Similar use of neighbouring reef areas in the Indian Ocean such as the Maldives have been applied for predicting maximum reef fish biomass in the WIO [100]. Second, within each ecoregion, we used reef sites that are protected from fishing in well managed protected areas that have been in place for at least a decade, i.e. effective in 2010 [101,102]. This resulted in between 2 and 17 protected sites per ecoregion (Table S10). Initial population values were calculated for each ecoregion, to account for natural variation in fish community structure and abundances across the WIO [6,103], by pooling sites from the reference locations and the protected areas (Table S10).

Table S10. Initial values for fish population densities for each ecoregion, derived from reference population density values from highly remote, uninhabited reference sites, Chagos Archipelago (central Indian Ocean) and Glorieuses Islands (WIO) and well protected sites within each ecoregion. For each ecoregion, the mean and standard deviation for well protected sites (named) is shown, and 'Initial values' is derived from those plus the values at the reference locations.

				Groupers (ind/ha)			***Parrotfish (ind/ha)***		
		# sites	**# yrs data**	**Mean**	**SD**	**cv**	**Mean**	**SD**	**cv**
Reference locations									
	Chagos	13	1	164.0	180.5	1.1	701.0	254.7	0.4
	Glorieuses	6	1	968.9	1295.0	1.3	413.3	245.4	0.6
Protected sites in each ecoregion									
N.Tanzania-Kenya	Kisite, Malindi, Watamu & Chumbe Island	16	16	145.6	353	2.4	631.5	782.5	1.2
N.Mozambique-S.Tanzania	Mafia, Mnazi Bay, Vamizi, Metundo	15	6	103.8	188.1	1.8	1088.3	923.2	0.8
Comoros	Mayotte and Moheli	17	9	347.5	519.2	1.5	399.6	286.8	0.7
West Madagascar	Velondriake NTZs	11	10	30.1	26.1	0.9	116.4	155.3	1.3
Seychelles North	Baie Ternay and Port Launay	6	13	110.4	134.2	1.2	984.8	781.1	0.8
Mascarene Islands	Etang Sale	2	15	106.5	345	3.2	N/A		
Delagoa	Bazaruto	8	1	4028.6	5117.3	1.3	243.8	182.1	0.7
Initial values									
N.Tanzania-Kenya		35	16	178.6	441.2	2.5	628.9	739.8	1.2
N.Mozambique-S.Tanzania		34	7	253.1	580.2	2.3	866.7	735.6	0.8
Comoros		36	11	378.8	649	1.7	492.7	300.1	0.6
West Madagascar		30	10	197.7	551.5	2.9	256.4	298.3	1.2
Seychelles North		25	13	167.8	377.3	2.2	915	728.6	0.8
Mascarene Islands		21	15	77.2	269.2	3.5	N/A		
Delagoa		27	2	1390.2	3067.5	2.2	501.6	304.3	0.6

6.1.2. Collapse thresholds

Collapse thresholds for each variable were identified by considering the lowest level the variable can decline to, before its function is considered minimal to absent, thereby causing severe biotic disruption to the reef ecosystem.

For hard coral cover a collapse threshold of 5% was selected (Table 1), based on multiple lines of evidence. On average, the hard coral cover threshold for coral reefs to be net accretive is 10% [32], providing an upper bound for this indicator. The accretive threshold of 10% was used for Criterion A to identify a 'functioning coral reef', whereas in Criterion D we are assessing trends in the coral populations with respect to algae and fish dynamics. Mortality events have frequently resulted in coral cover dropping below 10% for several years, for example in Kenya and Seychelles [15,21], but recovering back to higher levels, so coral cover < 10% may just be a temporary early successional stage in reef dynamics. Further, where coral cover is low after a mortality event but remaining cover is dominated by low algal turf forms, recovery of corals is not impeded, and the high turf/high herbivory conditions may favour coral recovery [81,84]. Some reefs have a naturally low coral cover of ~10%, such as in northern Kenya [45] and some reef flat systems. Previous coral reef RLE assessments have used very low thresholds of between 0-1% [42,44], though the most recent study in Colombia chose a value of 5% on very similar grounds to this study [104]. A threshold of 0-1% was determined to be too low, as a coral reef generally stops functioning as a reef before it loses all its coral cover. Finally, following the RLE guidelines an elicitation process at the 1st expert workshop for this study in March 2019 produced a range of recommendations from 21 participants from 2.5 to 9.1% coral cover (average lower and upper bounds) and average 'best estimate' of 6.2% coral cover.

For algae:coral ratio a collapse threshold of 0.83 was selected, based on the following: a reef was considered collapsed if the algae cover was 5 times the coral cover [105]. At this ratio, coral cover must be sufficiently low, comparable to the hard coral cover collapse threshold of 5% (therefore algae cover > 25%) and the limit of 10% for reef accretion (therefore algae cover > 50%). At these levels the fleshy algal cover (and other potentially dominant benthic taxa) would be sufficiently high to pose a significant barrier to coral recovery [82,106,107]. With this ratio, the maximum coral cover possible is 16.7%, in the unlikely case that the entire benthic substrate was made up of just coral and algae.

We are unaware of any published collapse thresholds for coral reef fishes, those used in the Meso American RLE were derived from a simulation model. Population collapse thresholds for fish trophic groups were calculated as a % of the initial value for each ecoregion, set at a maximum of 20% for groupers and 10% for parrotfish (Table S9). Productivity-biomass relationships for reef fishes suggest collapse in productivity and hence ecological function occurs between 25% and ~10% of unfished biomass [108]. The effects of removal of parrotfish is reported from Zanzibar by [109] who estimated a 90% reduction in the ecological function of scraping of algae when parrotfish biomass is reduced by 50%. Though there is also evidence that populations depleted to 10% of healthy populations can recover if effective fisheries management is put in place [110], such management is rare in the WIO [111,112]. We also compared results using a minimum collapse threshold of 1% that approximates a population of zero, unequivocally collapsed, as a sensitivity analysis to substantiate the results (as was done by Bland et al. (2017) in the Meso American RLE using simulations).

6.1.3. Bootstrapping relative severity and extent

Relative severity for all four variables was calculated with the standard equation, for each site in an ecoregion:

Relative severity (%) = (Observed decline / Maximum decline) × 100

where

Observed decline = Initial value – current value, and

Maximum decline = Initial value – Collapse value

The current value for each site was provided by the data on coral and algae cover and fish, for the period 2013-2019 (and the number of sites per ecoregion varied from 9 to 113), while the initial value for each site was provided by an ecoregion mean estimate with a standard deviation (Tables S7-10). To improve the accuracy and robustness of this method, we applied a bootstrap approach to resample from the distribution of initial values to calculate relative severity and extent of decline in each ecoregion. For each variable and each ecoregion, a single iteration comprised randomly extracting an initial value from a normal distribution defined by the ecoregion initial mean and standard deviation. This initial value is assigned to each site to calculate relative severity. The extent is calculated from the proportion of sites with relative severity above 30, 50 and 80. This was compared against the relative severity/extent matrix for Criterion D (Table 1) to identify three possible threat status classifications for that iteration. The highest (most severe) threat category was selected as the result for that iteration. For example, for hard coral cover in N. Mozambique-S Tanzania, with 98 sites (Table S7), if 50 of the sites (i.e. extent > 50%) had a relative severity of decline > 50% and 48/98 sites (or >30% extent) had relative severity 30% the classification would be VU. This process was repeated another 749 times, with the result being as tabulated in Table S12, where this would have been 1 of only 5 of the 750 iterations producing a VU result, 743 (99.1%) were LC, and 1 (0.1%) each were NT and CR.

A sensitivity analysis was run to determine the number of iterations at which the results for each ecoregion and variable stabilized (fig. S5). On the whole, results stabilised before 500 iterations, so we decided to run 750 iterations to ensure stable results.

Following guidance in the Red List Guidelines, if an area does not meet the criteria to be classified as Vulnerable it is considered Least Concern. However, to provide greater discrimination of an ecoregion approaching a vulnerable state, we defined the boundary for Near Threatened as within 10% of the Vulnerable extent limit for each relative severity level i.e. 27% extent for relative severity ≥ 80, 45% extent for relative severity ≥ 50 and 72% extent for relative severity ≥ 30.

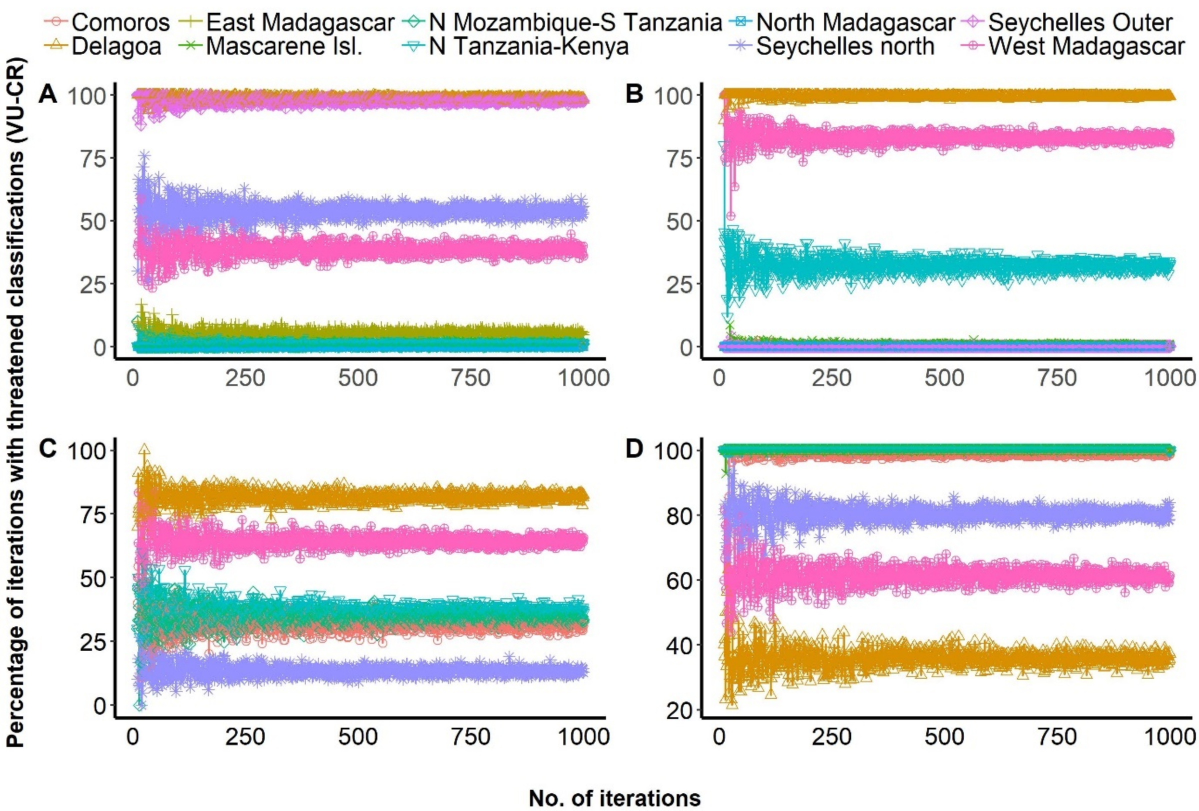

Figure S5. Repeated sampling trials to determine number of iterations to produce stable percentage of threatened classifications (VU-CR), from 10 to 1000 (x axis) for each of the following indicators. A: hard coral cover; B: algae:coral ratio; C: parrotfish; D: grouper.

The repeated sampling/bootstrap approach results in a spread of results across multiple risk categories (see Table S12-S14) for all four indicators. The spread was particularly wide for parrotfish and groupers due to wide abundance ranges typical in reef fish monitoring data (Table S9), as well as the wide standard deviations driving bootstrap sampling of Initial values (Table S10) for the 750 iterations. The RLE guidelines advise identification of a single risk category where possible, so we applied the following rules:

1. if > 80% of iterations fell within a single category, this category was selected;
2. if > 20% of iterations fell across multiple adjacent categories, these defined a range of categories;
3. some fish results spanned non-adjacent non-threatened and threatened categories (e.g. groupers in West Madagascar were 34.7% NT and 62.8% CR, Table S14). For these, we summed the non-threatened (LC+NT) and the threatened (VU+EN+CR) categories; if non-threatened was higher, then NT was selected, if threatened was higher, then VU was selected.

6.1.4. Synthesizing coral, algae and fish results

We determined the final risk level for Criterion D on the basis of an algorithm based on sequential interactions among the model compartments (fig. 2a, section 2.6). To test the implications of this algorithm (labelled 'Seq' for sequential) we tested it against two alternatives applying successively less structure. The three algorithms are summarized here:

- 'Seq' – starting with the coral risk level, the risk levels for algae, parrotfish then groupers are considered in sequence. If the risk status of the next compartment is the same as, or less than, that of the prior level, the current risk level is conserved. If the risk status of the next compartment is higher than that of the prior level, the current risk level is increased by one step, irrespective of the gap in status between the two (Table S2).
- 'Rank' – as with 'Seq' this algorithm starts with coral risk. The risk levels for the other three compartments are ranked from lowest to highest, then overall risk was stepped upwards from the initial value for corals, one step for each higher risk category. This applies an unordered ecosystem model that does not impose a sequential order as in our ecosystem model (fig. 2a), but like 'Seq' is sensitive to differential levels of risk across compartments.
- 'Max' - as is done in the RLE for combining results across Criteria, the most threatened risk level among biotic compartments is selected, irrespective of the others. Where Data Deficient is included as a category, i.e. where some compartments may not have sufficient data for assessment, this rule incorporates varying provision of data across compartments among units of assessment, as is the case in this study for parrotfish and grouper data among ecoregions.

The steps taken in this analysis were the following:

1. The number of possible levels for each compartment was set at 5 (i.e. LC, NT, VU, EN and CR, in increasing order), and number of compartments from 2 to 6;
2. Calculate all possible permutations (with repetition) of risk levels across all compartments;
3. Select final threat status for each permutation using all three algorithms;
4. Compare results of the three algorithms for concordance (i.e. equivalent, or 1, 2 or n levels different);
5. Repeat steps 1-4 for different number of compartments (2-6);

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6. Compare overall degree of concordance for the different number of compartments

Given there are 5 levels of risk (r), the possible combinations of risk among 'n' compartments is set by the formula for permutations with repetition, or r^n. Thus increasing the number of compartments increases dramatically the number of permutations, starting with 25 for 2 compartments, 625 for 4 compartments and 15,625 for 6 compartments (Table S11). Given the algorithms, it is axiomatic that the risk levels returned are Seq ≤ Rank ≤ Max.

We assessed the degree of concordance among the algorithms (i.e. if the ecosystem risk level was the same among algorithms, or 1, 2 or 3 risk levels different, see Table S11), the implications for interpretation of ecosystem risk, and the generality of this approach based on the number of ecosystem compartments. The latter factor is important as for any given coral reef ecosystem (or region): a) the dominant interactions affecting ecosystem state may vary from others, thus the type and number of compartments may vary, and b) available data to parametrize the compartments and their interactions may vary. Ideally, the algorithm chosen should be robust to these differences to facilitate comparison among assessments.

Table S11. Sample of permutations for the four-compartment model assessed in this study. The ecosystem risk levels given by the three algorithms (Seq, Rank and Max) are shown and the difference in risk level between the algorithms, for Max vs. Seq and Rank vs. Seq, based on the ordered set (LC < NT < VU < EN < CR). Cell entries are colour coded using the standard colours for the risk levels (see fig. 1) to facilitate reading the table.

Compartments				Algorithms			Difference (diff)	
Coral	Algae	Parrot	Grouper	Seq	Rank	Max	Max-Seq	Rank-Seq
LC	LC	LC	LC	LC	LC	LC	0	0
NT	LC	LC	LC	NT	NT	NT	0	0
EN	EN	EN	LC	EN	EN	EN	0	0
CR	EN	EN	LC	CR	CR	CR	0	0
EN	CR	EN	LC	CR	CR	CR	0	0
NT	LC	NT	NT	NT	NT	NT	0	0
NT	NT	EN	LC	VU	VU	EN	1	0
LC	VU	EN	LC	VU	VU	EN	1	0
LC	EN	EN	LC	VU	VU	EN	1	0
NT	CR	EN	LC	EN	EN	CR	1	0
LC	EN	EN	NT	VU	EN	EN	1	1
NT	CR	CR	EN	EN	CR	CR	1	1
VU	LC	LC	CR	EN	EN	CR	1	0
VU	NT	LC	CR	EN	EN	CR	1	0
NT	CR	VU	VU	VU	EN	CR	2	1
LC	CR	EN	VU	VU	EN	CR	2	1
LC	LC	CR	VU	VU	VU	CR	2	0
NT	LC	CR	VU	VU	EN	CR	2	1
LC	LC	LC	EN	NT	NT	EN	2	0
LC	CR	LC	LC	NT	NT	CR	3	0
LC	LC	CR	LC	NT	NT	CR	3	0
LC	LC	CR	NT	NT	VU	CR	3	1

6.2. Intermediate results

South Madagascar was Data Deficient for coral and algae in Criterion D, as no sites were monitored in this ecoregion, while 4 ecoregions were Data Deficient for groupers and 5 for parrotfish.

For coral cover alone (Table S12), six ecoregions were assessed as LC with >80% (4 with >94%) of iterations producing the same result – N Mozambique-S Tanzania, Comoros, East Madagascar, North Madagascar, N Tanzania-Kenya, Mascarene Islands. Two ecoregions were split across LC and VU – West Madagascar and Seychelles North, and the latter had a low occurrence (4.1%) of CR. Seychelles Outer and Delagoa were classified as VU with > 80% of iterations producing this status.

Across the region as a whole, decline in coral cover resulted in approximately 75% LC vs. 25% VU for the weighted average among ecoregions, 67% LC vs. 33% VU for the unweighted average, and 94.4% LC considering all sites equally across the whole region. As with criteria A and C, the weighted average is selected as best representing the condition of reefs across the region.

Table S12. Risk levels for each ecoregion based on coral cover, using a threshold for collapse of 5% coral cover. Cell values report the percentage of iterations (n=750) producing the given result.

Coral cover only		LC	NT	VU	EN	CR	Status
1	N.Tanzania-Kenya	92.8	5.6	0.7	0	0.9	LC
2	N.Mozambique-S.Tanzania	99.1	0.1	0.7	0	0.1	LC
3	Comoros	99.6	0.4	0	0	0	LC
4	West Madagascar	52	7.7	40.3	0	0	NT(LC-VU)
5	North Madagascar	99.3	0.1	0.4	0	0.1	LC
6	Seychelles Outer	0	2.1	97.7	0.1	0	VU
7	Seychelles North	33.5	12.1	50.7	0	3.7	NT(LC-VU)
8	Mascarene Islands	98.4	0	1.3	0	0.3	LC
9	East Madagascar	84.1	12.1	3.7	0	0	LC
10	South Madagascar		Data Deficient				DD
11	Delagoa	0.4	1.5	82.8	14.9	0.4	VU
WIO region							
Weighted average of ecoregions		75.7	3.6	20	0.1	0.5	NT(LC-VU)
Unweighted average of ecoregions		65.9	4.2	27.8	1.5	0.6	NT(LC-VU)
As one region		95.5	0.5	4	0	0	LC

For the algae:coral ratio (Table S13), seven ecoregions were assessed as LC (95-100 % of iterations). N Tanzania – Kenya had a broad spread of results spread across LC, NT and VU. West Madagascar and Delagoa were VU with 75 and 91 % of iterations, respectively. Across the whole region, the three approaches produced similar results to that for coral cover, with the weighted average giving ≈ 80 % of sites LC and 15 % VU.

Table S13. Risk levels for each ecoregion based on algae:coral ratio, using a threshold for collapse of 0.833. Cell values report the percentage of iterations (n=750) producing the given result.

Algae – coral ratio		LC	NT	VU	EN	CR	Status
1	N.Tanzania-Kenya	62.8	10.1	26	0.3	0.8	NT(LC-VU)
2	N.Mozambique-S.Tanzania	100	0	0	0	0	LC
3	Comoros	97.3	2.7	0	0	0	LC
4	West Madagascar	13.9	6.9	75.5	3.7	0	VU
5	North Madagascar	100	0	0	0	0	LC
6	Seychelles Outer	100	0	0	0	0	LC
7	Seychelles North	99.9	0	0	0	0.1	LC
8	Mascarene Islands	95.2	4.7	0.1	0	0	LC
9	East Madagascar	100	0	0	0	0	LC
10	South Madagascar		Data Deficient				DD

11	Delagoa	0.4	0.5	91.2	7.9	0	VU
WIO region							
	Weighted average of ecoregions	82.7	2.8	13.8	0.6	0.1	LC
	Unweighted average of ecoregions	77	2.5	19.3	1.2	0.1	NT(LC-VU)
	As one region	98.7	0.3	1.1	0	0	LC

Fish data were available for 6 (parrotfish) and 7 (groupers) of the 11 ecoregions. Parrotfish generally showed lower levels of threat, with three ecoregions assessed as NT overall, because, unlike the groupers, their highest (62–65%) iterations were LC but with notable scores in VU (North Mozambique-S. Tanzania) or CR (N. Tanzania-Kenya) (Table S14). In contrast, Seychelles North strongly returned LC (89% iterations), with a small proportion of CR (10% of iterations). However, West Madagascar and Delagoa had relatively high threatened (VU-CR) scores (66–83%), resulting in assignment of VU overall.

Groupers showed high levels of threat, with 31–63% of iterations giving a CR categorization across all seven ecoregions. Due to the high variance in Initial values (Table S10), results were spread across multiple risk categories, with four ecoregions having 49–57% of iterations in EN, and the other three ecoregions with 23-64% of iterations in LC and NT (Table S14). Final scores of VU were assigned to Seychelles North and West Madagascar because the proportion of scores in threatened categories (VU-CR, >62%) was greater than in non-threatened categories (LC-NT). In Delagoa, the opposite occurred, with 64.0% in LC and 36 % CR, therefore NT was assigned to this ecoregion.

Table S14. Risk levels for each ecoregion based on parrotfish and grouper abundance, using a threshold for collapse of 10% of initial values for parrotfish and 20% for groupers. Cell values report the percentage of iterations (n=750) producing the given result.

	Ecoregion	LC	NT	VU	EN	CR	Status
Parrotfish							
1	N.Tanzania-Kenya	61.9	2.4	14.1	0.3	21.3	NT (LC-CR)
2	N.Mozambique-S.Tanzania	65.7	0	17.7	0.3	14.8	NT (LC-VU)
3	Comoros	58.9	6.1	25.3	0.1	9.5	NT (LC-VU)
4	West Madagascar	21.2	12.9	13.3	28	24.5	VU (LC-CR)
7	Seychelles North	89.2	0.3	0.5	0	10.0	LC
11	Delagoa	16.7	0	61.2	14.4	7.7	VU
	WIO region						
	Weighted average of ecoregions	58.3	3.4	15.3	5.1	17.0	NT
	Unweighted average of ecoregions	52.3	3.6	22	7.2	14.9	NT (LC-VU)
	As one region	35.3	12.0	41.5	11.1	0.1	VU (LC-VU)
Grouper							
1	N.Tanzania-Kenya	0	0	1.1	56.4	42.5	EN-CR
2	N.Mozambique-S.Tanzania	0	0	0.5	48.9	50.5	EN-CR
3	Comoros	0	1.1	11.7	51.1	36.1	EN-CR
4	West Madagascar	2.5	34.7	0	0	62.8	VU(NT-CR)
7	Seychelles North	22.8	0.5	21.7	10.9	44.0	VU(LC-CR)
8	Mascarene Islands	0	0.1	12.3	56.9	30.7	EN-CR
11	Delagoa	64.0	0	0	0	36.0	NT(LC-CR)
	WIO region						
	Weighted average of ecoregions	3.8	5.7	5.0	38.8	47.6	EN-CR
	Unweighted average of ecoregions	12.8	5.2	6.8	32.0	43.2	VU(LC-CR)
	As one region	0	0	8.8	83.47	7.73	EN

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We tested these results at a more conservative 1% collapse threshold for both fish families. For parrotfish this resulted in similar overall scores to the maximum threshold results with 2 exceptions: Comoros dropped from NT to LC, and Delagoa from VU to NT. For groupers, the 1% collapse threshold resulted in LC scores (7-16%) where there were none previously, but conversely generally higher CR scores (32-87%). Overall, these balanced each other so the overall scores per ecoregion were unchanged.

The results for the four indicators in Criterion D are combined in Table S15. Taking the indicators in sequence, for coral cover alone, six ecoregions were LC, two were NT, and 2 were classified as VU. For the algae:coral ratio alone, seven ecoregions were assessed as LC, one as NT and two were VU. Because of high variation around population means for both current and initial values (Table S9, S10) fish indicators had a wide spread across threat categories (Table S14), resulting in broad threat classifications. For parrotfish three ecoregions were assessed as NT, while West Madagascar and Delagoa were assessed as VU and Comoros was LC. Grouper populations were more threatened, and somewhat reversed in their pattern: the three ecoregions assessed as NT for parrotfish were EN-CR for groupers, as was Mascarene Islands, while the two VU ecoregions for parrotfish were VU or NT for groupers.

The final Criterion D status (Tables 2, S15) was obtained using the stepwise ecosystem collapse algorithm based on multiple compartments (Table S2), with the process illustrated for selected ecoregions in Table S16.

Table S15. Combined results for hard coral, algae, herbivore and piscivore compartments assessed under Criterion D1. 'Overall' gives the final risk level following the structured ecosystem collapse model (Tables S2, S12-S14). The * denotes ecoregions for which the absence of data for the two fish compartments (hence Data Deficient for these compartments) may mask higher overall risk levels for Criterion D, resulting in lower confidence in these ecoregions risk status compared to other ecoregions.

		Coral	**Algae**	**Parrotfish**	**Groupers**	**Overall**
1	N.Tanzania-Kenya	LC	NT	NT	EN-CR	VU
2	N.Mozambique-S.Tanzania	LC	LC	NT	EN-CR	VU
3	Comoros	LC	LC	NT	EN-CR	VU
4	West Madagascar	NT	VU	VU	VU	VU
5	North Madagascar	LC	LC	DD	DD	LC*
6	Seychelles.Outer	VU	LC	DD	DD	VU*
7	Seychelles North	NT	LC	LC	VU	VU
8	Mascarene Islands	LC	LC	DD	EN-CR	NT
9	East Madagascar	LC	LC	DD	DD	LC*
10	South Madagascar	DD	DD	DD	DD	DD
11	Delagoa	VU	VU	VU	NT	VU
WIO region – weighted average		NT	LC	NT	EN-CR	VU

On the basis of changes in hard coral and fleshy algae cover, and parrotfish and grouper abundance over the past 50 years (criterion D), risk levels ranged widely from LC to CR (Table S15, fig. 3). For several ecoregions (North Tanzania-Kenya, North Mozambique-South Tanzania, Comoros), coral cover was LC, but ecosystem risk increased sequentially due to higher risk in the algae, parrotfish and/or grouper compartments. North and East Madagascar are known to experience high fishing pressure (SI7.3) but were Data Deficient for fish. Thus, their LC status, based only on coral and algae data, has limited confidence

(SI6.3, Table S19). Contrastingly, although Seychelles Outer was also Data Deficient for fish, its known low fishing pressure (Table S1) provides greater confidence in its VU status. Overall, for Criterion D, two ecoregions were LC (N Madagascar and E Madagascar), one was NT (Mascarene Islands), seven were VU and one was DD (South Madagascar, Tables 2, 3).

The algae data used potentially introduces errors, due to the breadth of algal forms combined into the variable 'fleshy algae', and may inflate risk where low-canopy turfs are included thus raising the level of algal cover, but without harming corals [81]. Nevertheless, the level of risk scored for algae was higher than that for corals in only two ecoregions (N.Tanzania-Kenya and West Madagascar), but in the latter the final risk level was not changed as neither fish compartment raised the risk level any higher (Table S15).

Table S16. Illustration of the sequential ecosystem collapse algorithm for biotic disruption, based on multiple compartments.

	Ecoregion and steps	Coral	Algae	Parrotfish	Groupers	Status
1	N.Tanzania...Kenya	LC	NT	NT	EN-CR	VU
	Compared to prior		*> LC*	*= NT*	*> NT*	
	Rationale	*Starting status*	*Increases one step*	*Remains the same*	*Increases one step*	
	Stepwise result:	*LC*	*NT*	*NT*	*VU*	
4	West Madagascar	NT	VU	VU	VU	VU
	Compared to prior		*> NT*	*= NT*	*> NT*	
	rationale	*Starting status*	*Increases one step*	*Remains the same*	*Remains the same*	
	Stepwise result:	*NT*	*VU*	*VU*	*VU*	
8	Mascarene Islands	LC	LC	DD	EN-CR	NT
	Compared to prior		*> LC*	-	*> LC*	
	rationale	*Starting status*	*Remains the same*	*No change*	*Increases one step*	
	Stepwise result:	*LC*	*LC*	*LC*	*NT*	

6.3. Methodological discussion

A weakness in our approach revolves around uncertainty in estimating initial values, given the gaps in data stretching back 50 years before the present. These gaps forced us to make assumptions that protected sites are well enforced and that remote reference sites are truly representative of healthy fish populations before 1970. To reduce dependency on precise initial values we applied a bootstrap approach to assign varied initial value to each site in an ecoregion. Particularly for fish, this high variance in both initial and current values resulted in a wide spread of risk categories (Table S14). Additional factors driving the large spread of results for fish include the variance inherent in reef fish monitoring data [113,114], high variation in fish populations between sites within ecoregions due to natural factors, especially in larger ecoregions [6], as well as variation in protection from fishing among sites within an ecoregion. All these factors may drive the spread of iterations to opposite ends of the risk categories scale, with some in LC and others in threatened and even CR categories (Table S14). Because of the high spread of categories within an ecoregion (e.g. LC to CR in N. Tanzania-Kenya for parrotfish) and the need to identify, where possible, a single category for an ecoregion, our result is necessarily a simplistic estimate of condition.

Additional factors affect accuracy of our results:

- An inherent bias is that initial selection of monitoring sites tends to prioritize high-value and thus high coral cover or high fish abundance sites. Thus these may only remain stable or decline. By comparison sites initially in a poor state that might not be prioritized as valuable for monitoring may subsequently improve in status, but this may not be captured.
- The number of sites within each ecoregion varied greatly, with low sample sizes below 15 or so affecting both benthic and fish datasets (see Tables S7-9).
- Complete absence of suitable data results in Data Deficient classification, for 1 ecoregion for coral and algae cover (South Madagascar), 4 ecoregions for groupers and 5 ecoregions for parrotfish. The multi-compartment model accommodates for some gaps in data, but because of the high risk level of groupers where data was available (in 4 of 7 ecoregions), their absence from ecoregions with the lower-risk classifications suggests these ecoregions may be more threatened than assessed here.

While corals are the foundation of the ecosystem model, their state did not drive the highest levels of risk. This was done by fish (particularly groupers), responding to fishing pressure, and by projected climate change, which will eventually impact directly on the corals. This emphasizes the inadequacy of percent coral cover alone as an adequate indicator of coral reef state, as reefs with reasonable coral cover may be closer to collapse on account of declining resilience in other compartments – such as fish, as analysed here, or through changes in composition of the coral community, which we could not assess. Particularly at smaller scales, other variables may have greater value as indicators of risk then the ones assessed at this higher spatial scale.

Concerns about increasing algal cover are growing in the WIO (e.g. [15], but this analysis indicates this is not yet a primary concern at ecoregional levels, compared to greater risk from declining fish populations and decreasing coral populations. The ecoregion with the highest concern from increasing algal populations is N.Tanzania-Kenya, which contains the highest human population density and therefore also pollution levels (it contains the principal cities of Dar es Salaam and Mombasa) across the WIO region. This indicator is also the least developed of those applied here, so the thresholds applied (Table 1) may require further work. The interaction between algae and corals, mediated by fish populations (which are under threat), coral die-offs (due to thermal stress) and increasing pollution and eutrophication levels, is the least well-supported interaction in the ecosystem model (fig. 2a), but emphasizes the importance of monitoring taxa that may become dominant if reefs transition to alternate states.

Groupers gave consistently higher levels of threat compared with parrotfish, which is expected because groupers are more vulnerable to overexploitation due to their life histories and behaviour [97], and fishing is widespread in the WIO. Also, parrotfish respond to degraded reefs in both negative and positive relationships [98,115], which is likely to produce variability in their results. A corollary of this is that absence of data for groupers can have a large impact on the final result, reducing confidence in the risk level estimated for ecoregions lacking grouper data (Table 3).

SUPPLEMENTARY INFORMATION

Coral reef ecosystems are vulnerable to critically endangered across an entire biogeographic province.

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6.3.1. Ecosystem collapse model and algorithms

Establishing what constitutes a collapsed ecosystem [43] is a primary question of this study (Section 2.6). While individual thresholds for collapse in individual compartments can be identified (Table 1), we assessed three algorithms for assigning overall risk levels across these – Seq, Rank and Max (SI6.1.4). At the same time, given our ecosystem model allows for differing numbers of compartments depending on dynamics of a particular region, and/or available data for each compartment, we assessed the effect of differing numbers of compartments on the final result; given we used four compartments (though for ecoregions Data Deficient for parrotfish and/or groupers we used less), we assessed from two to six compartments, based on potential for additional compartments (see fig. 2a, Table S17).

Table S17. Summary of the degree of concordance between the three algorithms for aggregating risk. 'Seq' – the sequential structured model described in Table S2 and applied in the assessment. 'Rank' is similar to 'Seq', starting with coral cover but the other three compartments were first ranked from lowest to highest before being sequentially considered. 'Max' follows a null approach, where the most threatened risk level was selected. See full descriptions in section 6.1.4. The results are compared across the three algorithms at multiple levels (sections a to e in the table) and for multiple numbers of ecosystem model compartments, from 2 to 6 (columns). Numbers within the table (apart from row "Total number of permutations") are all percentages. For example in row b) Seq and Rank gave the same result, and both were less than Max. 'Compartments' is shortened to 'comp' in the Comments column; 'diff' is the difference in final risk levels between the algorithms, see Table S11.

		Number of compartments ('comp')					**Comments**
		2	**3**	**4**	**5**	**6**	
Total number of permutations		25	125	**625**	3,125	15,625	
a) All approaches equal		**76**	**68**	**66**	**68**	**71**	Concordance is least for 4 comp (66%) but increases for fewer and more
b) Seq = Rank < Max		**24**	**27**	**22.1**	**15.1**	**9.7**	Seq and Rank give the same result, highest for 3 comp, lowest for 6. May be from 1 to 3 levels lower risk than Max, but magnitude of difference reduces with more compartments.
	1	12.0	16.0	**16.0**	12.3	8.5	
	2	8.0	9.6	**5.6**	2.7	1.2	
	b3	4.0	1.6	**0.5**	0.1	0.03	
c) Seq < Rank = Max		**n/a**	**2.4**	**6.6**	**12.2**	**15.7**	Disagreement between Seq and Rank/Max increases with more compartments. Almost all instances are 1 level lower, but may rarely be 2 levels lower for 5 and 6 comp.
	1		2.4	**6.6**	11.8	15.1	
	2				0.4	0.6	
d) Seq<Rank<Max		**n/a**	**2.4**	**5.0**	**4.7**	**3.4**	Different risk level among all three, maximal for 4 comp at 5%. Seq is always just 1 risk level lower than Rank. Rank is most frequently just 1 risk level lower than Max, rarely 2.
(Seq < Rank)	1		2.4	**5.0**	4.7	3.4	
(Rank < max)	1		1.6	**4.3**	4.3	3.2	
	2		0.8	**0.6**	0.4	0.2	
e) Equal to or within one risk level difference							
Seq <> Max		88.0	86.4	**89.0**	92.2	94.8	Proportion increases from 88% to 95% from 2 to 6 compartments. Very similar for 2-4 compartments

Seq <> Rank	100.0	100.0	**100.0**	99.6	99.4	Proportion is 100% from 2-4 compartments, decreases to 99.4% for 6 compartments
Rank <> Max	88.0	88.0	**93.3**	96.9	98.6	Proportion increases from 88% to 99% from2 to 6 compartments

Assessing the relevance of the differences between the algorithms can be done from two perspectives: a) computational, considering just the range of possible permutations (Table S17) and b) biological, considering the ecological relevance of the specific permutations. In the text below conclusions from the observations are highlighted in italics.

From a computational point of view, the following can be synthesized from Table S17, with a focus on the case in this study, of four ecological compartments (fig. 2a), and focusing on contrasting the extreme algorithms 'Seq' and 'Max':

- All three algorithms returning the same result is minimal for 4 compartments, at 66% of results (row a).
- For the 34% of cases where there is disagreement among the algorithms, for most of the cases (16%, row b and 6.6%, row c) there is only one category difference across them.
- For the 11% of cases that Seq returns a result 2 risk levels different from Max (row e), Seq and Rank are never more than 1 risk level different.

<u>*Conclusions*</u>

- *this 4-compartment case provides a 'worst case' model for performance of the sequential algorithm for the purposes of the RL approach. Unless an ecosystem has more than six compartments, the results presented here give confidence in the reliability of the structured collapse algorithm.*
- *When concordance among the algorithms is worst, Seq and Rank are never more than one level apart. Given the greater biological structure and meaning of Seq, it is selected as a more meaningful collapse model than Rank.*

If we accept that 1 risk level of difference is acceptable given the high levels of variation and uncertainty in nature, we focus investigation of the biological meaning of lack of concordance in the 11% of results where Seq is 2 or more levels less than Max (row f). We looked at all the permutations for which the difference was 2 or greater and found, of the 69 cases (11% of the total):

1. Coral was LC in 50 cases, and NT in 19 cases; i.e. the foundation of the reef model was non-threatened.
2. The result for Max was EN in 7 cases, and CR in 62 cases; i.e. at highest levels of risk. These 62 cases are a little under one third of the total of 216 cases that can result in CR (Table S17).
3. When coral was LC and Seq returned NT (14 cases), (i.e. overall risk level was not threatened), one of the other compartments was either EN or CR (threatened), and the other two compartments were LC or NT (non-threatened). Thus only one compartment was threatened.
4. When coral was LC and Seq returned VU (36 cases), two groups emerged:

- In one group (20 cases) one other compartment was also LC, and the other two could be NT, VU, EN or CR. In three cases two of the other compartments could be NT and one CR.
- In the other group (16 cases), one compartment may be NT or one or two may be VU, and one compartment CR
- Thus the compartments appeared relatively balanced between non-threatened and threatened, with one compartment pulling strongly to CR, but with a central tendency around VU.

5. When coral was NT and Seq gave a result of VU (19 cases), two groups emerged:
    - In one group (12 cases), two of the other compartments were LC or NT and only one was CR
    - In the other group (7 cases), one compartment one would be LC or NT, one VU and one CR

<u>*Conclusion:*</u>

- *Results 3, 4 and 5 are similar to the principle applied to the fish results where risk levels across the full range from LC - CR were obtained in 750 iterations (Table S14): when the predominant result was non-threatened, NT was selected as the aggregate level, versus where the predominant result was threatened, VU was selected.*
- *High levels of difference between the algorithms (diff = 2 or 3) are driven by a single compartment being CR, or EN, while all the others are at substantially lower risk levels, and corals were non-threatened (LC and NT).*

Finally, the primary decision is around whether a single compartment should give the whole ecosystem a rating of CR (see SI 2.6). Summing the number of permutations that give a final result from LC to CR for 2 to 6 compartments (Table S18) illustrates the permutational result that while there is only one permutation that allows an overall result of LC (all compartments must have a risk level of LC, so one out of 25 for two compartments and one out of 15,625 for six compartments), the more compartments there are the higher the proportion of permutations that result in higher overall risk levels. While the likelihood in reality of every permutation is not the same, this amplification of higher risk due to more compartments and the algorithm used further supports using the structured over the maximum algorithm for a result less biased by the mathematics towards high levels of risk.

Table S18. Percentage of permutations resulting in each level of overall risk, for 2 to 6 compartments. The number of permutations that may result in LC is one across all sets of compartments from 2 to 6, while the number that may result in CR increases from 6 for Sequential and Rank models for 2 compartments to 11,529 for Max with six compartments.

Model and risk level	**Number of Compartments**				
	2	**3**	**4**	**5**	**6**
			Cell values are %		
Sequential					
LC	4	0.8	0.2	0.03	0.01
NT	24	13	6	2.4	1.0
VU	24	29	25	18	13
EN	24	29	35	38	37
CR	24	29	35	41	49
Rank					
LC	4	0.8	0.2	0.03	0.01
NT	24	10	4	1.4	0.5

VU	24	30	20	12	7
EN	24	30	38	36	31
CR	24	30	38	50	62
Max					
LC	4	0.8	0.2	0.03	0.01
NT	12	6	2	1.0	0.4
VU	20	15	10	7	4
EN	28	30	28	25	22
CR	36	49	59	67	74
Total permutations	25	125	625	3,125	15,625

The generality of these results may provide a basis for applying this ecosystem collapse model and algorithm to other coral reef regions with differing ecosystem compartments, and to other ecosystems with a strong structuring compartment analogous to corals.

7. Broader considerations

7.1. Data limitations

The RLE method, based on five criteria and multiple sub-criteria allows for incorporation of disparate sources and types of data. This allows for identification of the most likely categorization of risk for an ecosystem within plausible bounds, and addressing sources of uncertainty identified by the RLE Guidelines (see section 2.8). The major sources of uncertainty are summarized in Table S19.

Table S19. Summary of the data limitations and sources of uncertainty of the study, distilled from relevant sections in the Supplementary Material and main text.

Issue	**Data limitations, sources of uncertainty**	**Resolution**	**Impact on analysis**	**Confidence**
Time period (50 years)	No primary data sources from 50 years ago.	Back-casting from oldest data sources, and reference sites.	Tendency to under-estimate historical conditions, though uncertain. May underestimate decline.	Moderate - High
		Used boot-strapping approach for variance in estimates	Enables variance in estimates, reduces reliance on mean value.	
Current values	Extensive data sources for the present, but with varying spatial distribution and quality.	Used statistical averaging, checked spatial distribution of values.	Small sample sizes and or aggregation give poor representation. Visual check of spatial distribution enables estimate of representativity.	Moderate - High
	Biases in site selection for monitoring; tendency to select 'best' sites, and discontinue collapsed sites.	Unable to address this in current dataset.	May underestimate the true distribution of sites which have declined below the collapse threshold.	Moderate - High
Limited threats assessed	Varied threats affect coral reefs, but most relatively localized, and lack of regionally representative data.	Impact of most threats known to be localized. Assessed the dominant threats at provincial	Little impact beyond local scale, little impact on ecoregional and regional patterns.	High

		scale - thermal stress and fishing.		
Criterion A	No accurate source of reef extent/ distribution in the past.	Used *in situ* coral cover and decline below 10% coral cover as threshold for declining coral reef extent (3.1).	Provides proxy estimate of reef ecosystem extent. Data redundancy with criterion D, though different thresholds associated with different interpretations of coral cover.	Moderate - High
	Threshold for reef decline at 10% coral cover	Conducted a sensitivity analysis (fig. S2)	Lower thresholds result in no threatened ecoregions below 7% coral cover, but relative risk among ecoregions is stable.	High
Criterion B	Uncertainty related to methods for Criterion B are standard to the RLE approach (Bland et al. 2015), and dependent on quality of habitat map.	Verification of the coral reef layer resulted in addition of minor reefs in East Madagascar and Delagoa, though concerns about 'missed' reefs in Delagoa persist, in Mozambique.	Strongly affect EOO and AOO, inconsistency with results from IUCN (unpubl.) are due to reefs added in this study, thus larger EOO and AOO, and lesser risk levels. Classifications in the coral reef layer used include areas which may not be true coral reefs, therefore over-estimating the presence/ distribution of reefs, and potentially underestimating the threat status.	High
Criterion C	SST projections based on downscaling of global climate models include inherent biases (van Hooidonk et al. 2016)	Use single published dataset, so internal consistency is high, whereas comparisons with current conditions or other models may be problematic. Used an intermediate rather than extreme RCP scenario	From a comparative perspective, relative vulnerability or risk should be reliable, but using these in absolute predictions (e.g. when a tipping point may occur).	Moderate
	Applying thermal stress assumptions from real conditions and variation (Liu 2012) to projected SST from global climate models may be problematic (van Hooidonk et al. 2016)			
	Seychelles North was assigned a risk level of LC, whereas the granitic islands, a major part of this ecoregion, were consistently more impacted by coral bleaching in 1998 and 2016 than most other parts of the WIO (Gudka et al. 2020).	The ecoregion contains both the granitic and the inner set of outer islands (7.3), so lesser vulnerability of the latter may influence the low risk level. Result may reflect limitations of downscaling from global models, and their inability to resolve smaller scale		

		phenomena. Use RCP 6.0 rather than 4.5.		
	Potential for acclimation and adaptation by corals and zooxanthellae to warming temperatures	Used the higher bleaching threshold of 12 DHW, currently associated with severe mortality of corals.	The degree of trait shift needed for corals to cope with the degree of warming expected is likely to be too high [116]	Moderate – High
Criterion D	Few sites in some ecoregions, minimum of 9-14 sites for some ecoregions	While low, this number of sites provides a reasonable estimate of variance.	Minor reduced confidence in these ecoregions.	High
	Data Deficiency for some ecoregions: 1for coral and coral-algae ratio, 4 for groupers, 5 for parrotfish.	Standard protocol for RLE for assigning Data Deficiency.	RLE approach is robust to Data Deficiency among Criteria.	High
		Collapse model designed to reduce influence of each compartment, but absence of data on groupers has a strong impact on final result.	Data Deficiency in some ecosystem compartments influences the final result, but is reduced using the structured model applied (2.6). Data deficiency of groupers in 2 ecoregions (East and North Madagascar) reduces confidence in the result due to known high levels of fishing (Table 3)	Moderate
	Limited sources of data for initial values.	Developed the boot-strapping approach to use variation in initial values and lessen the importance of the average value.	Provided a range of levels for initial values to complement the range of values of current data.	High
	Collapse model – required making assumptions about ecological interactions and whether to consider them as structured or not.	Permutations analysis enabled checking all possible outcomes of three collapse models, to assess concordance between them and ecological relevance of each model.	Established confidence in ±1 risk level precision and relative risk among ecoregions, and greatest ecological relevance of the structure/sequential model of assessing biotic risk.	Moderate - High
		Test dependence of result on # compart-ments and differential risk among them	Established confidence in ±1 risk level precision for applications with different numbers of compartments	Moderate
	Coral composition data not available, resulting in low risk based on total coral cover in the first step in the collapse model.	Could not address this limitation.	May result in incorrectly low assessment of risk in coral compartment, and thus lower overall assessment of risk.	Moderate
	Low- (<2 mm) and high- (3+ mm) canopy turfs in a single category, though their effects on corals very different	This was a limitation in input data. Improvements to monitoring methods may resolve this issue, but only for future data collection.	May overstate the competitive interactions between coral and algae. Nevertheless, other interactions on low turfs (e.g. grazing by urchins) disturb corals, and low-	High

			canopy turfs are closer in functional characteristics to other algae than to corals.	
	Aggregation of turf and macroalgae into a single indicator	This was a limitation in input data, due to different overlapping classifications being used for fleshy algal forms (see section 6.1 "Coral – algae ratio").	May overstate the competitive interactions between coral and algae. However this does preserved the basic dichotomy between hard orals and algae as dominant cover types.	High

Overall, these data limitations resulted in our assessing moderate to high confidence in the results of the analysis. Future assessments, in this region and in other regions, may be able to increase confidence by resolving further some or all of the limitations noted (Table S19).

In this assessment, the largest impacts of data limitation may be in the following areas:

a) Criterion C: improvements in the applicability of SST projections, use of current thermal stress thresholds to estimate risk in 50 years, and the low risk result for Seychelles North will strengthen future assessment. Actual levels of coral bleaching and mortality both within the WIO [14,21] and globally [18,52] and other projections of bleaching tipping points in the future [10,53,54,117] suggest that though our estimate may only have moderate confidence, actual risk levels may be higher than estimated here.

b) Criterion D: two factors may result in an underestimate of final levels of threat:
   - Grouper data was lacking from 4 ecoregions. As the most threatened biotic compartment, and at the end of the 'collapse chain' defined by our sequential algorithm (Table S2), the lack of grouper data may result in lower final risk levels in four ecoregions – North, East and South Madagascar, and Seychelles Outer (Table 3).
   - Coral cover was only available as an aggregate value, without any compositional detail. Detail on composition may indicate shifts from climate change 'losers', the fast growing corals that provide extensive structure and shelter on a reef, to climate change 'winners', many of which are slow growing and massive or encrusting in morphology, so provide less niche space for other species. Composition data may indicate higher risk levels for the coral compartment, and thus a high threat level at the start of the 'collapse chain' and thus a higher final level (Table S2).

Improving the above data limitations would increase confidence in the results, and might also result in higher risk levels assessed than those provided here. Internally, however, uncertainty as a result of data limitation may have little impact on the relative risk levels among ecoregions, within a Criterion. We thus have high confidence in the comparative assessment of risk among ecoregions.

7.2. Scale of assessment

Applying the RLE at the right scale and across the global extent of a biome or ecosystem is important [118] both from biological, and policy and management perspectives. The definition of an 'ecosystem' for the RLE includes the spatial aspect, in this study each ecoregion's coral reefs being viewed as a distinct coral reef ecosystem [2]. A relevant question is what is the range of scales across which the RLE approach can be applied? The global classification stretches across six levels, with level 1 being the whole planet, level 3 identifying coral reefs as a global ecosystem, and level 6 being the smallest.

Table S20. Pros and cons of applying the RLE to coral reefs at regional/ecoregional scales, based on past studies and this one.

Level	**RLE guidance and past practice**	**Our application**	
		Pros	**Cons**
Biogeographic province	Caribbean [42], Western Indian Ocean (this study)	• Simple result • Stable and consistent regional designation	• too large for Criterion B to be relevant • Large gaps in data masked by coverage from elsewhere within the region • Averaging of distinct characteristics of subregions/ecoregions • Scale too coarse for management
Ecoregion	Meso-American reef [44], Colombian Caribbean [104], this study (11 ecoregions)	• Differentiation among coral reef subregions addressed. • Consistency with ecoregional designations [3–5] • Consistency with genetic [119] and compositional findings [4,6,103].	• Scale too coarse for management • Poor fit to national/political boundaries (some countries have multiple ecoregions, some ecoregions split across countries) • Data deficient ecoregions arise due to gaps in historical sampling at these scales.

At the regional level, we found the assessment problematic (Table S20); there was significant variation in geographic representation, with heavy concentration of sites in some areas and few sites in others. This problem was reduced by reducing the scale of assessment to the ecoregional scale, where data gaps were explicitly flagged through DD classification. In addition, assessment at the ecoregion scale demonstrated variation in risk masked at the regional level.

Based on our experience, scales smaller than the smallest ecoregions we assessed may be too small for application of the RLE. First, Criterion B would be triggered almost automatically, as we found for our smaller ecoregions. Further, since coral reefs grow along a narrow depth-belt fringing the coastline, their actual area might be very small within any given EOO/AOO range. Second, subdividing the dataset at this scale would result in many smaller regions with insufficient or no study sites, thus many (likely a majority) would be assessed as DD, as was found in an assessment at this scale in Mozambique (IUCN Mozambique, unpubl.). Finally, given that this level in the global classification is intended as a 'bottom-up' approach for countries to complete the classification, there may be large inconsistencies and arbitrary differences among smaller units, making comparison among RLE assessments difficult.

From the above considerations, we find that the ecoregions, nested within the biogeographic province scale provide an appropriate scale for the RLE, useful for a) replicating consistently across the global extent of coral reef ecosystems, and b) applying the results consistently in regional and national policy processes for coral reef management. Correspondence of these scales to the global ecosystem classification [2] needs to be addressed, as it is unclear if they both fall within Level 4 (Biogeographic ecotype), or cross intermediate levels above or below (towards 'Global ecosystem type, or 'ecotype') this level .

Further downscaling is required to identify management responses to reef areas on a scale of several 10s to 100s of km, such as for marine protected areas and other OECMs. The Key Biodiversity Area (KBA) approach is under development targeting this scale of assessment [120], which may provide a natural linkage from broader scale RLEs to assess vulnerability, to finer scale KBA analysis to identify and plan actions.

7.3. Ecoregional results

This section summarizes some ecoregional discussion points important for next steps.

Seychelles outer islands – it is counter intuitive that this ecoregion is more endangered than the North Seychelles ecoregion, which contains the granitic islands most impacted by thermal stress and coral bleaching over the last two decades. This may reflect that North Seychelles includes some of the larger coral atoll systems (such as the Amirantes), thus masking the poor status of reefs in the granitic islands. Contrastingly, although this ecoregion was Data Deficient for fish, its known low fishing pressure (Table S1) provides greater confidence to its VU status.

South Madagascar is considered temperate, not tropical [11], hence its very limited area of coral reef (Table S1). In fact, as a warm-temperature region, this ecoregion's reefs may not meet full characteristics to be classed as a 'coral reef ecoregion' so its inclusion in this analysis is marginal (H. Razafindrainibe, pers. comm.). Its CR designation on the basis of future thermal stress may be counterintuitive as starting at minimum temperatures for coral reefs it may be expected to be a refuge for corals migrating from warmer northern latitudes. Also, because the thermal stress calculations are based on current conditions they may indicate greater stress under future warming, but don't address the possibility of such migration. Finally, the region had no coral reef data (so was DD for Criteria A and D), reflecting both the low cover of coral reefs, and it not being a priority for coral reef studies and monitoring.

North and East Madagascar are known to experience high fishing pressure (SI/Table S1) but were Data Deficient for fish, thus their LC status, based only on coral and algae data, has limited confidence. Contrastingly, although Seychelles Outer was also Data Deficient for fish, its known low fishing pressure (Table S1) provided greater confidence to its VU status. Both of these ecoregions also have small sample size, reducing confidence further.

SUPPLEMENTARY INFORMATION

Coral reef ecosystems are vulnerable to critically endangered across an entire biogeographic province.

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8. Data sources

This section acknowledges the contributions of all data providers and participants in this RLE process, covering primary data contributors, additional sources for current data, sources for initial value data and workshop participants.

8.1. Data contributors

The following contributed data to the regional coral reef dataset compiled by the Global Coral Reef Monitoring Network and used in regional [15] and global analyses (GCRMN, to be released in 2020):

Abigail Leadbeater (Blue Ventures, Madagascar); Alan Friedlander (Pristine Seas, Mozambique); Ali M Ussi (State University of Zanzibar, Tanzania); Alison Green (TNC, Mozambique), Aurelie Duhec (Marine Conservation Society Seychelles, Seychelles); Celine Miternique (Reef Conservation, Mauritius); Chloe Shute (Nature Seychelles, Seychelles); Colin Jackson (A Rocha Kenya); David Obura (CORDIO East Africa, Kenya, Mozambique, Comoros); Edward Mwamuye (EAWLS, Kenya); Eylem Elma (Tanzania); Hassan Kalombo (Fisheries, Tanga, Tanzania); Isabel Marques da Silva (Univ. Lurio, Pemba, Mozambique); Isabelle Ravinia (Seychelles National Parks Authority, Seychelles); January Ndagala (Marine Parks Reserves Unit, Tanzania); Jean Maharavo (Centre National de Recherches Océanographiques (CNRO), Madagascar); Jeanne WAGNER(Parc Naturel Marin de Mayotte, Mayote); Jennifer Olbers (Ezemvelo KZN Wildlife, South Africa); Josphine Mutiso (Kenya Wildlife Service, Kenya); Juliet Furaha (Kenya Marine and Fisheries Research Institute (KMFRI), Kenya); Juliette Damien (PRISM, Madagascar); Lautaro Alvarez (Frontier Madagascar, Madagascar); Linda Eggertsen (Stockholm University, Mozambique); Marcos A M Pereira (Centro Terra Viva - Estudos e Advocacia Ambiental, Mozambique); Mariliana Leotta (Green Islands Foundation, Seychelles); Marine Dedeken (Reunion NMR, Reunion); Melita Samoilys (CORDIO East Africa); Misbahou Mohamed (Dahari ONG, Comoros); Modesta Medard (WWF Tanzania); Mouchtadi Madi (Moheli Marine Park, Comoros); Mwaura Jelvas (Kenya Marine Fisheries Research Institute, Kenya); Nick Graham (Lancaster University, UK); Pádraig O'Grady (Madagascar Research and Conservation Institute (MRCI)); Pierre Andre-Adam (Islands Conservation Society, Seychelles); Ruben van Hooidonk (NOAA, USA); Said Ahamada (AIDE Comoros); Saleh Yahya (Institute of Marine Science, CARE-EARO, Tanzania); Sarah Freed (Portland State University, Comoros); Sean Porter (Oceanographic Research Institute, South Africa); Ulli Kloiber (Chumbe Island Coral Park, Tanzania)

8.2. Sources of current values

This section lists additional data sources to those provided by data contributors to the regional Global Coral Reef Monitoring Network dataset (section 8.1).

1. Costa A, Pereira M, Motta H, Schleyer M (2005) Status of the coral reefs of Mozambique:2004. In: Coral reef degradation in the Indian Ocean - Status Report 2005 (Souter, D., Linden, O., Eds). :54–60
2. Motta H, Pereira M a M, Gonçalves M, Schleyer MH, Ridgway T (2002) Mozambique Coral Reef. Coral Reef Monit Mozambique II 2000 Rep:31
3. Pereira MA, Gonçalves PMB (2004) Effects of the 2000 southern Mozambique floods on a marginal coral community: the case at Xai-Xai. African J Aquat Sci 29:113–116

4. Pereira M a M, Videira EJS, Costa ACD (2008) Update on coral reef activities in Mozambique (2004-2006). Biol Res:115–120
5. Pereira, M.A.M., C.M.M. Louro & R.S. Fernandes (2019). Reef monitoring in the Ponta do Ouro Partial Marine Reserve: 2018, 15 pp. Maputo, CTV
6. Louro, C. M. M., M. A. M Pereira, C. Litulo, R. S. Fernandes & T. I. F. C. Pereira (2018). Investigação e monitoria de espécies e ecossistemas nas áreas de conservação marinhas em Moçambique: Parque Nacional das Quirimbas 2017, 38 pp. Maputo, Centro Terra Viva.
7. Louro, C. M. M., M. A. M. Pereira, C. Litulo, M.H. Schleyer, R. S. Fernandes & T. I. F. C. Pereira (2018). Investigação e monitoria de espécies e ecossistemas nas áreas de conservação marinhas em Moçambique: Reserva Nacional do Pomene, 29 pp. Maputo, Centro Terra Viva.

8.3. Sources for initial values – coral and algae cover

Reference	DOI/URL	Coral/ algae
1. Ahamada, S., Bigot, L., Bijoux, J., Maharavo, J., Meunier, S., Moyne-Picard, M., & Paupiah, N. (2002). Status of coral reefs in the south west Indian Ocean island node: Comoros, Madagascar, Mauritius, Reunion and Seychelles. Status of the coral reefs of the world. Australian Institute of Marine Science, Townsville, 79-100.		Both
2. Chabanet, P., Ralambondrainy, H., Amanieu, M., Faure, G., & Galzin, R. (1997). Relationships between coral reef substrata and fish. Coral reefs, 16(2), 93-102.	https://doi.org/10.1007/s003380050063	Both
3. Costa, A., Pereira, M. A., Motta, H., & Schleyer, M (2005). Status of coral reefs of Mozambique: 2004. Coral reef degradation in the Indian Ocean: status report 2005, 54-60.		Coral
4. Graham, N. A., McClanahan, T. R., MacNeil, M. A., Wilson, S. K., Polunin, N. V., Jennings, S., ... & Bigot, L. (2008). Climate warming, marine protected areas and the ocean-scale integrity of coral reef ecosystems. PLoS one, 3(8), e3039.	https://doi.org/10.1371/journal.pone.0003039	Coral
5. Jennings, S., Grandcourt, E. M., & Polunin, N. V. C. (1995). The effects of fishing on the diversity, biomass and trophic structure of Seychelles' reef fish communities. Coral reefs, 14(4), 225-235.	https://doi.org/10.1007/BF00334346	Coral
6. McClanahan, T. R. (1994). Kenyan coral reef lagoon fish: effects of fishing, substrate complexity, and sea urchins. Coral reefs, 13(4), 231-241.	https://doi.org/10.1007/BF00303637	Coral
7. McClanahan, T. R., & Mutere, J. C. (1994). Coral and sea urchin assemblage structure and interrelationships in Kenyan reef lagoons. Hydrobiologia, 286(2), 109-124.	https://doi.org/10.1007/BF00008501	Coral
8. McClanahan, T. R., & Muthiga, N. A. (1988). Changes in Kenyan coral reef community structure and function due to exploitation. Hydrobiologia, 166(3), 269-276.	https://doi.org/10.1007/BF00008136	Coral
9. McClanahan, T. R., Ateweberhan, M., Muhando, C. A., Maina, J., & Mohammed, M. S. (2007). Effects of climate and	https://doi.org/10.1890/06-1182.1	Coral

seawater temperature variation on coral bleaching and mortality. Ecological Monographs, 77(4), 503-525.		
10. McClanahan, T. R., Muthiga, N. A., Maina, J., Kamukuru, A. T., & Yahya, S. A. (2009). Changes in northern Tanzania coral reefs during a period of increased fisheries management and climatic disturbance. Aquatic Conservation: Marine and Freshwater Ecosystems, 19(7), 758-771.	https://www.academia.edu/40137757/Changes_in_northern_Tanzania_coral_reefs_during_a_period_of_increased_fisheries_management_and_climatic_disturbance	Both
11. McClanahan, T., Muthiga, N., & Mangi, S. (2001). Coral and algal changes after the 1998 coral bleaching: interaction with reef management and herbivores on Kenyan reefs. Coral reefs, 19(4), 380-391.	https://doi.org/10.1007/s003380000133	Both
12. Muhando, C. (1999). Assessment of the extent of damage, socioeconomic effects, mitigation and recovery in Tanzania. Coral Reef Degradation in the Indian Ocean: Status Reports and Presentations, 43-46.		Both
13. Muhando, C. A., & Francis, J. (2000). The status of coral reefs in the Dar-Es-Salaam marine reserves system and the state of reefs in other marine protected areas of Tanzania. Status Report.	https://www.academia.edu/9525005/	Coral
14. Muhando, C. A., & Mohammed, M. S. (2002). Coral reef benthos and fisheries in Tanzania before and after the 1998 bleaching and mortality event. Western Indian Ocean J. Mar. Sci,1(1),43-52	https://www.wiomsa.org/download/muhando1(2).pdf	Both
15. Muthiga N, Abunge C, Kawaka J, Machaku R, Thoya P (2010) Training in Coral Reef and Fisheries Monitoring for Communities – Part 3.		Coral
16. Muthiga, N. A., & McClanahan, T. R. (1997). The effect of visitor use on the hard coral communities of the Kisite Marine Park, Kenya. In Proceedings of the 8th international coral reef symposium (Vol. 2, pp. 1879-1882).		Both
17. Ngusaru A.S. (2000). The Present State of Knowledge of Marine Science In Tanzania Synthesis Report	https://www.crc.uri.edu/download/2001_5047_TCMP_Knowledge.pdf	Coral
18. Obura, D.O. (1989). Coral reefs and sea urchins of the southern Kenya coast. B.A. thesis (Summa cum Laude), Harvard University, USA.		Both
19. Obura, DO. 1995. Environmental Stress and Life History Strategies, a Case Study of Corals and River Sediment from Malindi, Kenya. PhD Thesis. University of Miami, Miami, USA.		Both
20. Obura, D. (2002). Status of coral reefs in Eastern Africa: Kenya, Tanzania, Mozambique and South Africa.		Coral
21. Obura, D. (2002). Status of coral reefs in Kiunga Marine reserve, Kenya. Coral reef degradation in the Indian ocean: Status Report 2002, 47-54.		Coral

22. Obura, D. O. (1999). Status Report Kenya. Coral reef degradation in the Indian Ocean: Status reports and project presentations 1999., 33-36.		Both
23. Quod, J. P., & Bigot, L. (2000). Coral bleaching in the Indian Ocean islands: Ecological consequences and recovery in Madagascar, Comoros, Mayotte and Reunion. Coral reef degradation in the Indian Ocean, 108-113.	https://www.researchgate.net/publication/259441412_Coral_bleaching_in_the_Indian_Ocean_islands_Ecological_consequences_and_recovery_in_Madagascar_Comoros_Mayotte_and_Reunion	Both
24. Quod, J. P., Dahalani, Y., Bigot, L., Nicet, J. B., Ahamada, S., & Maharavo, J. (2002). Status of coral reefs at Réunion, Mayotte and Madagascar. Coral Reef Degradation in the Indian Ocean, 185.		Coral
25. Riegl, B., Schleyer, M. H., Cook, P. J., & Branch, G. M. (1995). Structure of Africa's southernmost coral communities. Bulletin of Marine Science, 56(2), 676-691.	https://www.ingentaconnect.com/content/umrsmas/bullmar/1995/00000056/00000002/art00022	Coral
26. Spalding, M. D., & Jarvis, G. E. (2002). The impact of the 1998 coral mortality on reef fish communities in the Seychelles. Marine Pollution Bulletin, 44(4), 309-321.	10.1016/S0025-326X(01)00281-8	Both

8.4. Workshop participants

The following attended the inception (12-15 March 2019) and/or validation (21-22 January 2020) workshops of this coral reef Red List of Ecosystems process, in Mombasa, Kenya:

Armindo Araman (Administração Nacional das Áreas de Conservação (ANAC), Mozambique); Chiranjiwa Naidoo Paupiah (Ministry of Ocean Economy, Marine Resources, Fisheries and Shipping, Mauritius); David Keith (University of New South Wales, Australia); David Obura (CORDIO East Africa, Kenya); Edward Ouko (Regional Centre for Mapping Resource For Development, Kenya); Elisa Cavoto (IUCN, Switzerland); Francisco Zivane (IIP, Mozambique); Hajanirina RAZAFINDRAINIBE (Centre National de Recherches Océanographiques (CNRO), Madagascar); Hamadi Mwamlavya (The Nature Conservancy, Kenya); Hassan Mohamed (World Wildlife Fund Kenya); Ihando Andrianjafy (Ministry of Environment, Madagascar); Isabelle Ravinia (Seychelles National Parks Authority, Seychelles); James Mbugua (CORDIO East Africa, Kenya); January Ndagala (Marine Parks Reserves Unit, Tanzania); Japhet Moroa (Coast Development Authority, Kenya); Jessica Rowland (Deakin University, Australia); John Komakoma (Marine Parks and Reserve Unit, Tanzania); Josphine Mutiso (Kenya Wildlife Service); Judith Nyunja (Kenya Wildlife Service); Juliet Furaha (Kenya Marine and Fisheries Research Institute); Julius Edward (National Environment Management Council, Tanzania); Majambo Gamoyo (CORDIO East Africa, Kenya); Marcos Valderrabano (IUCN, Spain); Melita Samoilys (CORDIO East Africa, Kenya); Mercy Amai (National Environmental Management Authority, Kenya); Mishal Gudka (CORDIO East Africa, Kenya); Moses Egaru (IUCN, Uganda); Mouchtadi

Madi (Moheli Marine Park, Comoros); Mwaura Jelvas (Kenya Marine Fisheries Research Institute); Naseeba Sidat (Wildlife Conservation Society, Mozambique); Nassur Ahmada Mroimana (Ministry of Environment, Comoros); Nima Raghunathan (IUCN, UK); Patrick Kimani (Coastal Marine Resource Development, Kenya); Randall Mabwa (CORDIO East Africa); Roberto Komeno (Reef Doctor, Madagascar); Ronan Roche (Bangor University, UK); Sabrina Meunier (Shoals Rodrigues, Mauritius); Said Ahamada (AIDE Comoros); Saleh Yahya (Institute of Marine Science, Tanzania); Sean Porter (Oceanographic Research Institute, South Africa); Stephen Katua (National Environmental Management Authority, Kenya); Sushma Mattan-Moorgawa (University of Mauritius); Swaleh Ali (CORDIO East Africa); Victor Dunga (South African National Biodiversity Institute (SANBI)); Victoria Kio (Coast Development Authority, Kenya).

9. References

1. Bland, L. M., Keith, D. A., Miller, R. M., Murray, N. J. & Rodríguez, J. P. *Guidelines for the application of IUCN Red List of ecosystems categories and criteria*. (IUCN International Union for Conservation of Nature, 2015). doi:10.2305/IUCN.CH.2016.RLE.1.en.
2. Keith, D. A., Ferrer-Paris, J. R., NICHOLSON, E., Bishop, M. J. & Polidoro, B. A. *The IUCN Global Ecosystem Typology v1.0: Descriptive profiles for Biomes and Ecosystem Functional Groups*. 229 (2020).
3. Spalding, M. D. *et al.* Marine ecoregions of the world: a bioregionalization of coastal and shelf areas. *BioScience* **57**, 573–583 (2007).
4. Obura, D. O. The Diversity and Biogeography of Western Indian Ocean Reef-Building Corals. *PLoS ONE* **7**, e45013 (2012).
5. Veron, J., Stafford-Smith, M. G., Devantier, L. M. & Turak, E. Overview of distribution patterns of zooxanthellate Scleractinia. *Frontiers in Marine Science* **1**, 1–19 (2015).
6. Samoilys, M. A., Halford, A. & Osuka, K. Disentangling drivers of the abundance of coral reef fishes in the Western Indian Ocean. *Ecol Evol* **9**, 4149–4167 (2019).
7. IUCN. *Assessing the Risk of Ecosystem collapse in Madagascar, using the IUCN Red List of Ecosystems*. 33 (unpubl).
8. WIOMER. *Réseau des Aires Marines Protégées des pays de la COI—Priorisation et Stratégie de WIOMER*. 158 (2009).
9. Andrefouet, S. *et al.* Global assessment of modern coral reef extent and diversity for regional science and management applications: a view from space. in 1732–1745 (2006).
10. Beyer, H. L. *et al.* Risk-sensitive planning for conserving coral reefs under rapid climate change. *Conservation Letters* **109**, e12587 (2018).
11. Bouchet, P., Perez, T. & Le Gall, L. ATIMO VATAE cruise,Antea R/V. (2010) doi:10.17600/10110040.
12. McClanahan, T. R., Sheppard, C. R. C. & Obura, D. O. *Coral Reefs of the Indian Ocean: Their Ecology and Conservation*. (Oxford University Press, 2000).
13. Obura, D. O. Coral and biogenic reefs. in *Regional State of the Coast report, Western Indian Ocean* 71–82 (WIOMSA, UNEP Nairobi Convention, 2015).
14. Ateweberhan, M., McClanahan, T. R., Graham, N. A. J. & Sheppard, C. R. C. Episodic heterogeneous decline and recovery of coral cover in the Indian Ocean. *Coral Reefs* **30**, 739–752 (2011).
15. Obura, D. *et al. Coral reef status report for the Western Indian Ocean.* 171 (2017).

16. IPCC. *Summary for Policymakers. In: IPCC Special Report on the Ocean and Cryosphere in a Changing Climate. 2019. [H.- O. Pörtner, D.C. Roberts, V. Masson-Delmotte, P. Zhai, M. Tignor, E. Poloczanska, K. Mintenbeck, M. Nicolai, A. Okem, J. Petzold, B. Rama, N. Weyer (eds.)].* https://www.ipcc.ch/site/assets/uploads/sites/3/2019/12/SROCC_FullReport_FINAL.pdf (2019).
17. Hoegh-Guldberg, O. *et al.* Coral Reefs Under Rapid Climate Change and Ocean Acidification. *Science* **318**, 1737–1742 (2007).
18. Eakin, C. M., Sweatman, H. P. A. & Brainard, R. E. The 2014–2017 global-scale coral bleaching event: insights and impacts. *Coral Reefs* **38**, 539–545 (2019).
19. Sheppard, C., Davy, S., Pilling, G. & Graham, N. *The Biology of Coral Reefs*. (Oxford University Press, 2017).
20. Stuart-Smith, R. D. *et al.* Integrating abundance and functional traits reveals new global hotspots of fish diversity. *Nature* **501**, 539–542 (2013).
21. Gudka, M. *et al.* Participatory reporting of the 2016 bleaching event in the Western Indian Ocean. *Coral Reefs* **39**, 1–11 (2020).
22. Obura, D. O. An Indian Ocean centre of origin revisited: Palaeogene and Neogene influences defining a biogeographic realm. *J. Biogeogr.* **43**, 229–242 (2016).
23. Kulbicki, M. *et al.* Global Biogeography of Reef Fishes: A Hierarchical Quantitative Delineation of Regions. *PLoS ONE* **8**, e81847 (2013).
24. Parravicini, V. *et al.* Global patterns and predictors of tropical reef fish species richness. *Ecography* **36**, 1254–1262 (2013).
25. Barber, P. H. & Meyer, C. P. Pluralism explains diversity in the Coral Triangle." (2015): 258. in *Ecology of fishes on coral reefs* (ed. Mora, C.) 258 (2015).
26. Allen, G. R. Conservation hotspots of biodiversity and endemism for Indo-Pacific coral reef fishes. *Aquatic Conservation: Marine and Freshwater Ecosystems* **18**, 541–556 (2008).
27. DiBattista, J. D. *et al.* After continents divide: comparative phylogeography of reef fishes from the Red Sea and Indian Ocean. *Journal of Biogeography* **40**, 1170–1181 (2013).
28. Groeneveld, J. The Western Indian Ocean as a source of food. in *Regional State of the Coast report, Western Indian Ocean* (ed. Paula, J.) 261–270 (WIOMSA, UNEP Nairobi Convention, 2015).
29. Samoilys, M. A. *et al. A rapid assessment of coral reefs at Metundo Island, Cabo Delgado, northern Mozambique*. 41 pp (2015).
30. Séré, M. G., Chabanet, P., Turquet, J., Quod, J. P. & Schleyer, M. H. Identification and prevalence of coral diseases on three Western Indian Ocean coral reefs. *Diseases of Aquatic Organisms* **114**, 249–261 (2015).
31. Goreau, T., McClanahan, T., Hayes, R. & Strong, A. Conservation of Coral Reefs after the 1998 Global Bleaching Event. *Conservation Biology* **14**, 5–15 (2000).
32. Perry, C. T. & Alvarez-Filip, L. Changing geo-ecological functions of coral reefs in the Anthropocene. *Funct Ecol* 1365-2435.13247 (2018) doi:10.1111/1365-2435.13247.
33. Smith, J. E. *et al.* Re-evaluating the health of coral reef communities: baselines and evidence for human impacts across the central Pacific. *Proceedings of the Royal Society B: Biological Sciences* **283**, 20151985 (2016).
34. Mumby, P. J., Steneck, R. S. & Hastings, A. Evidence for and against the existence of alternate attractors on coral reefs. *Oikos* **122**, 481–491 (2013).

35. Bellwood, D. R., Hughes, T. P., Folke, C. & Nyström, M. Confronting the coral reef crisis. *Nature* **429**, 827–833 (2004).
36. Mumby, P. J., Hastings, A. & Edwards, H. J. Thresholds and the resilience of Caribbean coral reefs. *Nature* **450**, 98–101 (2007).
37. Ainsworth, C. H. & Mumby, Peter. J. Coral-algal phase shifts alter fish communities and reduce fisheries production. *Glob Change Biol* **21**, 165–172 (2015).
38. Wittebolle, L. *et al.* Initial community evenness favours functionality under selective stress. *Nature* **458**, 623–626 (2009).
39. Bellwood, D. R. *et al.* Coral reef conservation in the Anthropocene: Confronting spatial mismatches and prioritizing functions. *Biological Conservation* **236**, 604–615 (2019).
40. GCRMN. *GCRMN Implementation and Governance Plan*. 21 (2019).
41. Obura, D. O. *et al.* Coral Reef Monitoring, Reef Assessment Technologies, and Ecosystem-Based Management. *Front. Mar. Sci.* **6**, 580 (2019).
42. Keith, D. A. *et al.* Scientific Foundations for an IUCN Red List of Ecosystems. *PLoS ONE* **8**, e62111 (2013).
43. Bland, L. M. *et al.* Developing a standardized definition of ecosystem collapse for risk assessment. *Frontiers in Ecology and the Environment* **16**, 29–36 (2018).
44. Bland, L. M. *et al.* Using multiple lines of evidence to assess the risk of ecosystem collapse. *Proc. R. Soc. B* **284**, 20170660 (2017).
45. Samoilys, M. A. Abundance and species richness of coral reef fish on the Kenyan Coast: The effects of protective management and fishing. in *Proc. Int.Coral Reef Symp.* vol. 2 261–266 (1988).
46. Church, J. & Obura, D. Sustaining Coral Reef Ecosystems and their Fisheries in the Kiunga Marine National Reserve, Lamu, Kenya. in *Proc. 10th International Coral Reef Symposium, Okinawa* 10 (2004).
47. Schleyer, M. & Celliers, L. Biodiversity on the marginal coral reefs of South Africa: what does the future hold? *Zoologische Verhandelingen* **345**, 387–400 (2003).
48. Pereira, M. A. M., Louro, C. M. M. & Fernandes, R. S. Reef Monitoring in the Ponta Do Ouro Partial Marine Reserve: 2018. 18 (2018).
49. West, J. M. & Salm, R. V. Resistance and Resilience to Coral Bleaching: Implications for Coral Reef Conservation and Management. *Conservation Biology* **17**, 956–967 (2003).
50. Obura, D. O. Resilience and climate change: lessons from coral reefs and bleaching in the Western Indian Ocean. *Estuarine, Coastal and Shelf Science* **63**, 353–372 (2005).
51. Hoegh-Guldberg, O., Poloczanska, E. S., Skirving, W. & Dove, S. Coral Reef Ecosystems under Climate Change and Ocean Acidification. *Frontiers in Marine Science* **4**, 321 (2017).
52. Hughes, T. P. *et al.* Coral reefs in the Anthropocene. *Nature* **546**, 82–90 (2017).
53. Hughes, T. P. *et al.* Spatial and temporal patterns of mass bleaching of corals in the Anthropocene. *Science* **359**, 80–83 (2018).
54. van Hooidonk, R. *et al.* Local-scale projections of coralreef futures and implications ofthe Paris Agreement. *Scientific Reports* 1–8 (2016) doi:10.1038/srep39666.
55. Liu, G. NOAA Coral Reef Watch's Decision Support System for Coral Reef Management. *12th International Coral Reef Symposium* 1–5 (2012).
56. Hoegh-Guldberg, O. Coral reefs and projections of future change. in *Coral Health and Disease. Eds: Eugene Rosenberg and Yossi Loya* 463–484 (Springer Verlag, 2004).

57. van Hooidonk, R., Maynard, J. A. & Planes, S. Temporary refugia for coral reefs in a warming world. *Nature Climate Change* **3**, 1–4 (2013).
58. Reynolds, R. W., Rayner, N. A., Smith, T. M., Stokes, D. C. & Wang, W. An Improved In Situ and Satellite SST Analysis for Climate. *J. Climate* **15**, 1609–1625 (2002).
59. Donner, S. D., Heron, S. F. & Skirving, W. J. Future Scenarios: a Review of Modelling Efforts to Predict the Future of Coral Reefs in an Era of Climate Change. in 159–173 (Springer Berlin Heidelberg, 2009). doi:10.1007/978-3-540-69775-6_10.
60. Climate Data Operators (CDO). Climate Data Operators (CDO). *Climate Data Operators* https://code.mpimet.mpg.de/projects/cdo/wiki/Cdo#Documentation (2020).
61. Darling, E. S. *et al.* Social–environmental drivers inform strategic management of coral reefs in the Anthropocene. *Nature Ecology & Evolution* **3**, 1341–1350 (2019).
62. Hughes, T. P. *et al.* Global warming transforms coral reef assemblages. *Nature* **556**, 492–496 (2018).
63. van Hooidonk, R. & Huber, M. Quantifying the quality of coral bleaching predictions. *Coral Reefs* **28**, 579–587 (2009).
64. Coles, S. L. *et al.* Evidence of acclimatization or adaptation in Hawaiian corals to higher ocean temperatures. *PeerJ* **6**, e5347 (2018).
65. Langdon, C., Albright, R., Baker, A. C. & Jones, P. Two threatened Caribbean coral species have contrasting responses to combined temperature and acidification stress. *Limnology and Oceanography* **63**, 2450–2464 (2018).
66. Obura, D. O., Bigot, L. & Benzoni, F. Coral responses to a repeat bleaching event in Mayotte in 2010. *PeerJ* **6**, e5305 (2018).
67. Hausfather, Z. & Peters, G. P. RCP8.5 is a problematic scenario for near-term emissions. *Proc Natl Acad Sci USA* **117**, 27791–27792 (2020).
68. Hausfather, Z. & Peters, G. P. Emissions – the 'business as usual' story is misleading. *Nature* **577**, 618–620 (2020).
69. Schwalm, C. R., Glendon, S. & Duffy, P. B. Reply to Hausfather and Peters: RCP8.5 is neither problematic nor misleading. *Proc Natl Acad Sci USA* **117**, 27793–27794 (2020).
70. Schwalm, C. R., Glendon, S. & Duffy, P. B. RCP8.5 tracks cumulative CO_2 emissions. *Proc Natl Acad Sci USA* **117**, 19656–19657 (2020).
71. United Nations Environment Programme. *The emissions gap report 2020*. (2020).
72. Ahamada, S. & Bijoux, J. P. Status of the Coral Reefs of the south-west Indian Ocean Island States: Comoros, Madagascar, Mauritius, Reunion, Seychelles. in *Status of coral reefs of the World* (eds. Bruce Cauvin, Annelise Hagan, Alasdair Harris, Meera Koonjul, Sabrina Meunier, Jean-Pascal Quod, C. & Wilkinson, C.) 105–118 (GCRMN and AIMS, 2008).
73. van Hooidonk, R., Maynard, J. A., Liu, Y. & Lee, S.-K. Downscaled projections of Caribbean coral bleaching that can inform conservation planning. *Global Change Biology* **21**, 3389–3401 (2015).
74. Graham, N. A. J. *et al.* Climate Warming, Marine Protected Areas and the Ocean-Scale Integrity of Coral Reef Ecosystems. *PLoS ONE* **3**, e3039 (2008).
75. Celliers, L. & Schleyer, M. H. Coral bleaching on high-latitude marginal reefs at Sodwana Bay, South Africa. *Marine Pollution Bulletin* **44**, 1380–1387 (2002).
76. Porter, S. N. & Schleyer, M. H. Long-term dynamics of a high-latitude coral reef community at Sodwana Bay, South Africa. *Coral Reefs* 1–14 (2017) doi:10.1007/s00338-016-1531-z.

77. Bay, R. A., Rose, N. H., Logan, C. A. & Palumbi, S. R. Genomic models predict successful coral adaptation if future ocean warming rates are reduced. *Science Advances* **3**, e1701413 (2017).
78. Kubicek, A., Breckling, B., Hoegh-Guldberg, O. & Reuter, H. Climate change drives trait-shifts in coral reef communities. *Scientific Reports* **9**, 3721 (2019).
79. Wilkinson, C. *Status of coral reefs of the world: 2008*. 304 (2008).
80. Obura, D. *Coral Reef Monitoring in the Southwest Indian Ocean*. 35 (2013).
81. Steneck, R. S., Mumby, P. J., MacDonald, C., Rasher, D. B. & Stoyle, G. Attenuating effects of ecosystem management on coral reefs. *Sci. Adv.* **4**, eaao5493 (2018).
82. McCook, L., Jompa, J. & Diaz-Pulido, G. Competition between corals and algae on coral reefs: a review of evidence and mechanisms. *Coral Reefs* **19**, 400–417 (2001).
83. Smith, J. E. indirect effects of algae on coral. 1–11 (2006).
84. Arnold, S., Steneck, R. & Mumby, P. Running the gauntlet: inhibitory effects of algal turfs on the processes of coral recruitment. *Mar. Ecol. Prog. Ser.* **414**, 91–105 (2010).
85. McClanahan, T. R. & Shafir, S. H. Causes and consequences of sea urchin abundance and diversity in Kenyan coral reef lagoons. *Oecologia* **83**, 362–370 (1990).
86. Bajjouk, T. *et al.* Detection of changes in shallow coral reefs status: Towards a spatial approach using hyperspectral and multispectral data. *Ecological Indicators* **96**, 174–191 (2019).
87. Bellwood, D. R., Streit, R. P., Brandl, S. J. & Tebbett, S. B. The meaning of the term 'function' in ecology: a coral reef perspective. *Funct Ecol* 1365-2435.13265 (2018) doi:10.1111/1365-2435.13265.
88. Bonaldo, R., Hoey, A. & Bellwood, D. The Ecosystem Roles of Parrotfishes on Tropical Reefs. in *Oceanography and Marine Biology* (eds. Hughes, R., Hughes, D. & Smith, I.) vol. 20141169 81–132 (CRC Press, 2014).
89. Bellwood, D. R., Schultz, Ortwin, Siqueira, A. C. & Cowman, P. F. A review of the fossil record of the Labridae. *Ann. Naturhist. Mus. Wien, Serie A* **121**, 125–193 (2019).
90. Sandin, S. & Zgliczynski, B. Inverted trophic pyramids. *Ecology of fishes on coral reefs* 247–251 (2015).
91. Hussey, N. E. *et al.* Stable isotopes and elasmobranchs: tissue types, methods, applications and assumptions. *Journal of Fish Biology* **80**, 1449–1484 (2012).
92. Boaden, A. E. & Kingsford, M. . J. Predators drive community structure in coral reef fish assemblages. *Ecosphere* **6**, 1–33 (2015).
93. Frisch, A. J., Ireland, M. & Baker, R. Trophic ecology of large predatory reef fishes: energy pathways, trophic level, and implications for fisheries in a changing climate. *Mar Biol* **161**, 61–73 (2014).
94. Heithaus, M. consequences of predator decline. 1–9 (2008) doi:10.1016/j.tree.2008.01.003.
95. McCauley, D. J. *et al.* Assessing the effects of large mobile predators on ecosystem connectivity. *Ecological Applications* **22**, 1711–1717 (2012).
96. Sadovy de Mitcheson, Y. *et al.* Fishing groupers towards extinction: a global assessment of threats and extinction risks in a billion dollar fishery: Fishing groupers to extinction. *Fish Fish* **14**, 119–136 (2013).
97. Sadovy de Mitcheson, Y. J. *et al.* Valuable but vulnerable: Over-fishing and under-management continue to threaten groupers so what now? *Marine Policy* **116**, 103909 (2020).

98. Samoilys, M., Roche, R., Koldewey, H. & Turner, J. Patterns in reef fish assemblages: Insights from the Chagos Archipelago. *PLoS ONE* **13**, e0191448 (2018).
99. Chabanet, P. *et al.* Coral reef monitoring in the Iles Eparses, Mozambique Channel (2011–2013). *Acta Oecologica* **72**, 62–71 (2016).
100. McClanahan, T. R. *et al.* Identifying Reefs of Hope and Hopeful Actions: Contextualizing Environmental, Ecological, and Social Parameters to Respond Effectively to Climate Change. *Conservation Biology* **23**, 662–671 (2009).
101. da Silva, I. M., Hill, N., Shimadzu, H., Soares, A. M. V. M. & Dornelas, M. Spillover Effects of a Community-Managed Marine Reserve. *PLoS ONE* **10**, e0111774 (2015).
102. Gilchrist, H., Rocliffe, S., Anderson, L. G. & Gough, C. L. A. Reef fish biomass recovery within community-managed no take zones. *Ocean & Coastal Management* **192**, 105210 (2020).
103. Osuka, K., Kochzius, M., Vanreusel, A., Obura, D. O. & Samoilys, M. Linkage between fish functional groups and coral reef benthic habitat composition in the Western Indian Ocean. *Journal of the Marine Biological Association of the United Kingdom* 1–14 (2016) doi:10.1017/S0025315416001399.
104. Uribe, E. S., Luna-Acosta, A. & Etter, A. Red List of Ecosystems: Risk assessment of coral ecosystems in the Colombian Caribbean. *Ocean & Coastal Management* **199**, 105416 (2021).
105. Bruno, J. F., Sweatman, H., Precht, W. F., Selig, E. R. & Schutte, V. G. W. Assessing evidence of phase shifts from coral to macroalgal dominance on coral reefs. *Ecology* **90**, 1478–1484 (2009).
106. Roth, F. *et al.* Coral reef degradation affects the potential for reef recovery after disturbance. *Marine Environmental Research* **142**, 48–58 (2018).
107. Johns, K. A. *et al.* Macroalgal feedbacks and substrate properties maintain a coral reef regime shift. *Ecosphere* **9**, e02349 (2018).
108. Morais, R. A., Connolly, S. R. & Bellwood, D. R. Human exploitation shapes productivity–biomass relationships on coral reefs. *Glob Change Biol* **26**, 1295–1305 (2020).
109. Lokrantz, J., Nyström, M., Thyresson, M. & Johansson, C. The non-linear relationship between body size and function in parrotfishes. *Coral Reefs* **27**, 967–974 (2008).
110. Condy, M., Cinner, J. E., McClanahan, T. R. & Bellwood, D. R. Projections of the impacts of gear-modification on the recovery of fish catches and ecosystem function in an impoverished fishery: ECOLOGICAL AND ECONOMIC IMPLICATIONS OF ESCAPE SLOTS IN FISHING TRAPS. *Aquatic Conserv: Mar. Freshw. Ecosyst.* **25**, 396–410 (2015).
111. Fulanda, B., Ohtomi, J., Mueni, E. & Kimani, E. Fishery trends, resource-use and management system in the Ungwana Bay fishery Kenya. *Ocean & Coastal Management* **54**, 401–414 (2011).
112. Samoilys, M. A., Osuka, K., Maina, G. W. & Obura, D. O. Artisanal fisheries on Kenya's coral reefs: Decadal trends reveal management needs. *Fisheries Research* **186**, 177–191 (2017).
113. Samoilys, M. A. & Carlos, G. Determining methods of underwater visual census for estimating the abundance of coral reef fishes. *Environmental Biology of Fishes* **57**, 289–304 (2000).

114. Usseglio, P. Quantifying reef fishes: bias in observational approaches. in *Ecology of Fishes on Coral Reefs* (ed. Mora, C.) 270–273 (Cambridge University Press, 2015). doi:10.1017/CBO9781316105412.035.
115. Russ, G. R., Questel, S.-L. A., Rizzari, J. R. & Alcala, A. C. The parrotfish–coral relationship: refuting the ubiquity of a prevailing paradigm. *Marine Biology* 1–18 (2015) doi:10.1007/s00227-015-2728-3.
116. Camp, E. F. *et al.* The Future of Coral Reefs Subject to Rapid Climate Change: Lessons from Natural Extreme Environments. *Front. Mar. Sci.* **5**, 4 (2018).
117. Sheppard, C. R. Predicted recurrences of mass coral mortality in the Indian Ocean. *Nature* **425**, 294–297 (2003).
118. Bland, L. M. *et al.* Impacts of the IUCN Red List of Ecosystems on conservation policy and practice. *CONSERVATION LETTERS* **12:e12666.**, 8 (2019).
119. Obura, D. O. *et al.* The Northern Mozambique Channel. in *World Seas: an Environmental Evaluation* 75–99 (Elsevier, 2019). doi:10.1016/B978-0-08-100853-9.00003-8.
120. IUCN. *A Global Standard for the Identification of Key Biodiversity Areas*. https://portals.iucn.org/library/sites/library/files/documents/2016-048.pdf (2016).