Supporting Information

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SI Methods

Study Sites. Coral reef benthic habitat and associated mobile fauna (reef fishes and urchins) were surveyed at 157 individual sites across the Indian Ocean, spanning ≈35° latitude and ≈52° longitude. Reefs were surveyed in Kenya, Tanzania, Mozambique, Mayotte, Madagascar, Reunion, Mauritius, Seychelles, and the Maldives. Data were collected from 1988 to 2009, resulting in a database of 335 site–time combinations (Table S1). The total physical area of coral reef surveyed was ≈550 km^2. Of the 335 individual surveys, 109 were in no-take fisheries closures (i.e., no-take marine protected areas), 109 were in areas with restrictions on gear use [for example, a ban on the use of beach seine nets (1)], and 117 were in areas fully open to all fishing gears. All reef surveys were conducted on the reef flat or shallow reef slope (≤10 m depth).

Eight types of ecological data were collected and evaluated: biomass of reef fish (kg/ha); species richness of reef fish (number per 500 m^2); biomass of herbivorous reef fish (scarids, acanthurids, and siganids; kg/ha); biomass of reef fish capable of preying on sea urchins (balistids and labrids >30 cm; kg/ha); sea urchin biomass (kg/ha); a predation index based on tethered sea urchin assays (proportion of urchins preyed upon); cover of macroalgae (%); hard coral cover (%); and cover of all calcifying organisms (hard corals, coralline, and calcareous algae; %). The number of sites with data on each of these ecological categories varied (Table S1). These data were used to calculate metrics that were indicative of key reef states and processes. This included changes in herbivory and nutrient inputs (2–4), rates of organic and calcium production (5), predation (6, 7), biodiversity (8, 9), mode of herbivory (10, 11), erosion and accretion (12, 13), secondary production (14), structural complexity (15), and recruitment (16).

The abundance of all diurnally active, noncryptic, reef-associated fishes was quantified at each site and their size estimated using underwater visual census techniques. In Kenya, Tanzania, Mozambique, Mayotte, Madagascar, Mauritius, and Maldives fish were counted within three to five 5 × 100-m belt transects (17). In Seychelles, fish were surveyed within 16-point counts, each with a 7-m radius (18). In Reunion, 4 × 50-m belt transects were used to count fish (19). In all methods, observers avoided double counting by surveying larger mobile species first and disregarding individuals that left the survey boundary and then reentered. There may be small amounts of variation associated with different survey techniques; however, studies comparing methods have found little difference between strip transects and point counts in estimating fish abundance (20, 21). Experienced observers collected all fish data, and the variability associated with experienced observer bias has been found to be small (22), including among four observers used here (23, 24).

These data were standardized by area surveyed and used for the calculation of total biomass (wet weight), herbivorous fish biomass, sea urchin predator biomass, and fish species richness. Biomass was estimated from individual fish-length data using length–weight relationships for species or families (25, 26). Biomass was calculated for fish >10 cm total length, because abundance of smaller-bodied fish is typically underestimated using visual census (27). Pomacentrids were excluded from biomass estimates because the size cutoff meant that certain species had been excluded in some locations. Large subsets of the current dataset have been standardized in this way for previous studies (15, 28, 29).

The cover of benthic categories (hard coral cover, macroalgae, and other calcifying organisms) was quantified visually in Seychelles sites, using point counts of photographed transects at Reunion and line intercept transects at other locations. These methods have been shown to yield comparable results (30, 31).

Sea urchin biomass was quantified by multiplying sea urchin population densities by an average wet body weight for each species (32). Spatial variation in the weight of urchin conspecifics is negligible, although *Echinometra* and *Diadema spp.* are often smaller on reefs closed to fishing (32, 33). Consequently, biomass of urchins from these genera was calculated from average weights in protected or unprotected sites.

Predation was estimated from survival rates of tethered urchins, *Echinometra mathaei*. At each site the test of 30–60 *E. mathaei* was pierced with a hypodermic needle and tethered with monofilament line to a transect line that was secured to the reef. An index of predation was calculated from the proportion of tethered urchins that were still alive after 24 h. An index value of 1 is indicative of very high predation and values of 0 very low predation (34) and is strongly related to observed predation by large balistids and labrids (35).

Unfished reef fish biomass (B_o) was calculated from surveys of marine parks and unfished reefs in the region (28). The compilation included 35 estimates of biomass from parks older than 15 y and larger than 5 km^2 and 12 estimates from unfished Maldivian coral reefs. The mean biomass from these studies was 1,192 kg/ha ± 378 (SD) (n = 47, SE = 55). Although unfished biomass may vary in other parts of the world (36), and this will influence threshold values, these estimates were most representative of unfished biomass for shallow-water Western Indian Ocean (WIO) reefs where there has been a long history of disturbance from fishing.

We use the concept of B_{MMSY}, which is the biomass that allows for maximum sustainable yield from the community; this represents a multispecies extension of the well-known sustainable yield concept for single stocks (37, 38). There is insufficient empirical information on tropical coral reef species to establish a single point estimate, but we can hypothesize a range from deductive reasoning of population growth models. Sigmoid population growth will have maximum rate of change at half the leveling value or 0.5 B_0, whereas populations with initial rapid exponential growth, followed by saturation, will have maximum rates of change at lower levels. Maximum rates of change could, mathematically or theoretically, occur at values close to zero biomass, but biological limits to reproduction and growth will ensure that these values are considerably above zero. This is supported by various stock-recruitment models for temperate species (37). Consequently, we chose 0.25 B_0 to represent a likely lower limit to populations with exponential saturation behavior. This range was created to determine where a likely MMSY window would lie in relationship to ecological switch points.

The log of national human population per kilometer of coastline (RPC) was used to order the average biomass by management type among the nine study countries (Fig. 2). This index represents a relative level of human resource pressure that was negatively related to average fished biomass (FB) among nations (FB = 2,900.1–320.3 × RPC; R^2 = 0.47), which explained approximately half of the country-level differences in observed fish biomass.

Data Analysis. Eight ecological metrics for each site in each year were modeled using a statistical distribution and link function appropriate to the response type (39). The distribution families were chosen a priori, given the nature of the response. For instance, percentage values were modeled using a 0–100 binomial model, whereas unbounded responses such as ratios were assumed to be normally distributed. These choices were examined a posteriori for model fit. Total fishable biomass was used as the predictor variable, because it is the variable most sensitive to management action. A strong correlation (0.88) between total fishable biomass and herbivorous fish biomass suggests that the role of herbivory is inherent in these models, because biomass of herbivorous fish is directly related to the process of herbivory. The eight metrics were (*i*) the proportion of macroalgal substrate (binomial distribution; logit link); (*ii*) the ratio (log-odds of observing) macroalgal to hard coral substratum (multinomial distribution; log link); (*iii*) the predation index for urchins (binomial distribution; logit link); (*iv*) the total species richness for fish (negative-binomial distribution; log link); (*v*) the proportion of herbivorous fish in the community (binomial distribution; logit link); (*vi*) the total biomass of urchins present (rounded to nearest kg; negative-binomial distribution; log link); (*vii*) the proportion of calcifying substratum (hard coral plus calcifying algae; binomial distribution; logit link); and (*viii*) the proportion of hard coral substratum (binomial distribution; logit link). Binomial and multinomial models for proportions were structured by multiplying the proportion observed (between 0 and 1) by 100 and taking the number of "trials" to be 100.

To identify and quantify the presence of potential thresholds between fishable biomass and each of the ecological metrics, we compared model fit among four candidate linear models: a null model, linear model, switch-point model, and piecewise linear model. The null (intercept-only) model was simply:

$$\begin{gathered} y_i = \beta_0 + \varepsilon_i \\ \varepsilon_i \sim N(0, \tau) \end{gathered}, \qquad \textbf{[S1]}$$

describing the mean (β_0) and variance (τ) of each metric (y_i); distribution families and link functions were used to modify the normal distribution in Eq. S1 as defined above. The linear model included a slope (β_1) for the relationship between each metric and biomass (BIO):

$$\begin{gathered} y_i = \beta_0 + \beta_1 * BIO_i + \varepsilon_i \\ \varepsilon_i \sim N(0, \tau) \end{gathered}. \qquad \textbf{[S2]}$$

The switch-point model estimated the location of a change (sp) in average metric values across the range of observed biomass:

$$\begin{gathered} y_i = \left\{ \begin{array}{l} \beta_l + \varepsilon_{il}, \ \ if \ \ BIO < sp \\ \beta_u + \varepsilon_{iu}, \ \ if \ \ BIO > sp \end{array} \right\} \\ \varepsilon_{il} \sim N(0, \tau_l) \\ \varepsilon_{iu} \sim N(0, \tau_u) \end{gathered}, \qquad \textbf{[S3]}$$

with independent lower (β_l, τ_l) and upper (β_u, τ_u) means and variances separated by a model-estimated switch-point (sp) that indicated the location of a threshold. Finally, we included a piecewise linear model:

$$\begin{gathered} y_i = \left\{ \begin{array}{c} \beta_0 + \beta_1 * BIO + \varepsilon_{il}, \ \ if \ \ BIO < sp \\ \beta_0 + (\beta_1 - \beta_2) * sp + \beta_2 * BIO + \varepsilon_{iu}, \ \ if \ \ BIO > sp \end{array} \right\} \\ \varepsilon_{il} \sim N(0, \tau_l) \\ \varepsilon_{iu} \sim N(0, \tau_u) \end{gathered} \qquad \textbf{[S4]}$$

that allowed independent lower and upper variances for two line segments connected at a model-estimated switch-point (threshold; sp). Note that the structure of Eq. S3 and Eq. S4 can provide support for a switch-point change in the mean model, variance, or both, when present in the data.

Because the complexity of some models (e.g., multinomial switch-point) precluded conventional methods, all models were run on each ecological metric in a Bayesian framework using the PyMC (40) package for the Python programming language (41). Variance priors were G(0.0001, 0.0001); switch-point priors were U(0, 1,500); and linear parameter priors were N(0.0, 0.0001), specified with precision. Relative model support was assessed using the deviance information criterion (DIC) (42), whereby lower-valued DIC scores provided support for one model over another. Model fit was assessed using Bayesian *P* values (43), whereby discrepancies less than 0.05 or greater than 0.95 between expected and model-simulated values relative to the observed data were taken to indicate a lack of model fit (44). Metrics with DIC-based support for the presence of a threshold related to fishable biomass and sufficient model fit were retained for subsequent inference.

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Table S1. Number of observations per site included for each ecological variable modeled by country, year range of sampling, and management

Country	Management	Biomass	Species richness	Urchin predators	Herbivorous fish biomass	Urchin biomass	Urchin predation index	Macroalgal cover	Coral cover	Calcifying organism cover
Kenya	Closure	53	18	53	53	51	50	52	52	49
1988–2008	Restricted	20	9	20	20	20	15	18	18	18
	Fished	44	8	44	44	39	39	44	44	40
Madagascar	Closure	6	2	2	6	5	0	5	5	5
1998–2009	Restricted	7	5	5	7	6	0	8	8	3
	Fished	32	28	28	32	24	0	25	25	13
Maldives	Closure	0	0	0	0	0	0	0	0	0
2005	Restricted	12	12	12	12	0	0	11	11	11
	Fished	0	0	0	0	0	0	0	0	0
Mauritius	Closure	2	2	2	2	2	2	2	2	2
2004	Restricted	5	4	5	5	5	5	5	5	5
	Fished	7	5	7	7	6	6	6	6	6
Mayotte	Closure	7	7	7	7	0	0	5	5	0
2009	Restricted	0	0	0	0	0	0	0	0	0
	Fished	8	8	8	8	0	0	8	8	0
Mozambique	Closure	6	6	6	6	6	0	6	6	6
2008–2009	Restricted	0	0	0	0	0	0	0	0	0
	Fished	8	8	8	8	5	0	5	5	5
Reunion	Closure	0	0	0	0	0	0	0	0	0
2005–2006	Restricted	16	16	16	16	16	2	16	16	16
	Fished	6	6	6	6	6	0	6	6	6
Seychelles	Closure	27	0	27	27	0	0	27	27	0
1994–2008	Restricted	36	0	36	36	0	0	36	36	0
	Fished	0	0	0	0	0	0	0	0	0
Tanzania	Closure	8	8	8	8	8	8	8	8	8
1996–2008	Restricted	13	13	13	13	13	8	13	13	9
	Fished	12	12	12	12	12	8	12	12	12
Grand total		335	177	325	335	224	143	318	318	214

Table S2. Model selection results for candidate threshold models

Indicator	Null	Linear	Switchpoint	Piecewise	ΔDIC
Percentage macroalgal cover	5,398.95	5,310.95	5,288.93	**5,203.08**	85.85
Macroalgae-hard coral ratio	1,269.50	1,052.33	**974.97**	1,002.38	27.41
Urchin predation index	7,915.64	3,539.08	3,400.43	**3,342.94**	57.39
Fish species richness	1,520.16	1,360.49	1,284.00	**1,237.64**	46.36
Percent herbivorous fish	5,825.35	5,761.99	5,352.60	**5,248.51**	104.09
Urchin biomass	3,532.00	3,501.05	3,471.86	**3,462.74**	9.12
Percent calcifying substrate	4,125.29	3,696.77	3,512.03	**3,378.03**	133.00
Percent hard coral	6,196.20	6,112.61	6,142.77	**5,979.22**	133.39

Models in bold have the greatest DIC-based support; ΔDIC values indicate DIC distance between the top two ranked models; ΔDIC values >2 indicate substantial support for the lower-valued model. Macroalgae-hard coral ratio is the log-odds of observing macroalgae vs. hard coral.

Table S3. Posterior threshold estimates for highest-ranked models

Indicator	Mean	Median	95% UI
Percentage macroalgal cover	1,132	1,126	1,023–1,269
Macroalgae-hard coral ratio	851	851	835–856
Urchin predation index	642	635	607–730
Fish species richness	261	258	217–319
Percentage herbivorous fish	228	236	117–271
Urchin biomass	158	165	122–173
Percentage calcifying substrate	144	144	136–156
Percentage hard coral	91	91	79–101

Results include 95% uncertainty intervals (95% UI) based on highest posterior density for the threshold parameter of the top DIC-ranked model for each indicator. Macroalgae-hard coral ratio is the log-odds of observing macroalgae vs. hard coral.