



Supplementary Materials for

Meeting fisheries, ecosystem function, and biodiversity goals in a human-dominated world

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Materials and Methods

Scales of data

Our data were organized at four spatial scales: survey (n=4399), reef site (n=1797), reef cluster (n=734), and nation/state (n=41).

- i) surveys were our smallest scale of data – see details about survey methods below.
- ii) reef sites were aggregations of replicate surveys within a few hundred meters. There were an average of 2.4 replicate surveys per reef site.
- iii) reef clusters- We clustered reef sites together that were within 4 km of each other, and used the centroid of these reef clusters to estimate certain social and environmental covariates (Table S3). To make reef clusters, we first estimated the linear distance between all reef sites, then used a hierarchical analysis with the complete-linkage clustering technique based on the maximum distance between reef sites. We set the cut-off at 4 km to select mutually exclusive reef clusters where reef sites cannot be more distant than 4 km. The choice of 4km was informed by a 3-year study of the spatial movement patterns of artisanal coral reef fishers, corresponding to the highest density of fishing activities on reefs based on GPS-derived effort density maps of artisanal coral reef fishing activities (24). This clustering analysis was carried out using the R functions ‘hclust’ and ‘cutree’, resulting in an average of 2.7 reef sites/reef cluster.
- iv) Nation/state (nation, state, or territory). A larger scale in our analysis was ‘nation/state’, which are jurisdictions that generally correspond to individual nations (but could also include states, territories, overseas regions, or extremely remote areas

within a state such as the Hawaii or the British Indian Ocean Territory; Table S1), within which reef clusters and reef sites were nested for analysis.

Reef fish survey methods

Estimates were based on instantaneous visual counts from 4399 surveys collected from 1798 tropical reef sites (i.e., within 23.5 latitude degrees). All surveys used standard belt-transects, distance sampling, or point-counts, and were conducted between 2004 and 2013. For each site, habitat type (i.e., slope, crest, flat, lagoon/back reef), depth range (i.e., 0-4m, 4-10m and >10m) and total sampling area were recorded. Where data from multiple years were available for a single reef site, we included only data from the year closest to 2010. Within each survey area, reef-associated fishes were identified to species level, abundance counted, and total length (TL) estimated, with the exception of one data provider who measured biomass at the family level.

As part of our standardization process, we:

- i) Retained families that were consistently studied and were above a minimum size cut-off. Thus, we retained counts of >10cm non-cryptic reef fishes from families that are resident on the reef (Table S4).
- ii) Directly accounted for depth, survey method, survey area, and habitat as covariates in the model.

Key ecological metrics

We then used these surveys to calculate three key reef fish ecological metrics:

i) *Biomass of reef fish above 20 cm*. We calculated total biomass of fish above 20 cm (TL) on each reef site (n= 1798) using standard published species-level length-weight relationship parameters or those available on FishBase (25). When length-weight relationship parameters were not available for a species, we used the parameters for a closely related species or genus. Included families are specified in Table S4.

ii). *Parrotfish Scraping Potential*. Scraping rates (area grazed per minute) for parrotfishes at each reef site (n=1662) were calculated as the product of parrotfish fish density, feeding rate, and bite dimension (area) (26). Size-specific feeding rates were derived from best-fit regressions of bite rate (bites min⁻¹) and fish length ([TL], cm) for each species or closely related congener. Bite rates for Indo-Pacific parrotfishes were quantified at three locations (Great Barrier Reef, Australia; Indonesia; and the Red Sea) during which TL was estimated and the number of bites on different benthic substrata (primarily epilithic algal matrix and live corals) recorded and converted to bites min⁻¹. Individual fish were followed for a minimum of 3-minutes and 19-126 individuals (mean = 41 individuals) were observed per species. These values were supplemented with published length-feeding rate relationships, including for Atlantic parrotfishes (reviewed in (27)). Size-specific bite dimensions (mm²) were taken from the literature (26, 28–30).

iii). *Trait Diversity*. Trait diversity was computed for each reef site, considered as a local fish community (n=1662). First, we used the trait database on tropical reef fishes from Mouillot et al. (31) to describe species traits. The six traits considered were: (1) size (observed length of each individual fish) coded using 5 ordered categories: 10-15 cm, 15.1-30 cm, 30.1-50 cm, 50.1-80 cm, >80 cm; (2) mobility coded using 3 ordered

categories: sedentary, mobile within a reef, and mobile between reefs; (3) period of activity coded using 3 ordered categories: diurnal, both diurnal and nocturnal, and nocturnal; (4) schooling coded using 5 ordered categories: solitary, paired, or living in small (3-20 individuals), medium (20-50 individuals), or large groups (>50 groups) ; (5) vertical position in the water column coded using 3 ordered categories: benthic, bentho-pelagic, and pelagic; (6) diet coded using 7 trophic categories: herbivorous-detritivorous, macro-algal herbivorous, invertivorous targeting sessile invertebrates, invertivorous targeting mobile invertebrates, planktivorous, piscivorous, and omnivorous, (i.e. fishes that feed on both vegetal and animal material). Since all traits were categorical, species with identical traits were grouped into entities. We then computed the Gower distance between all pairs of entities. Finally, for each fish community we computed trait-diversity using the Chao's $FD_{q=1}$ index (7):

$$FD_{q=1} = \exp \left(- \sum_{i=1}^s p_i \times \log \left(1 - \sum_{i \neq j} \frac{1 - \min(d_{ij}, mD)}{mD} \times p_j \right) \right)$$

where p_i and p_j are the respective relative biomasses of the two entities i and j in the community, d_{ij} is the Gower distance between entities i and j , mD is the average of all Gower distances between the entities present in the global pool of species. This index is expressed as an equivalent numbers of species (7). Hence, it is minimal and equals 1 when all biomass is supported by the same entity (i.e. when one species is ultra-dominant or when all species have the same trait values) and it is maximal and equals the number of

species when all species pairs have dissimilarities higher than the average dissimilarity in the global species pool and equal biomasses.

We used species-level data to calculate parrotfish scraping potential and trait diversity. Thus, data from the one provider who only recorded family level data were not used in those response variables.

Social and environmental potential drivers

1. Management: For each reef site, we determined if it was: i) unfished- whether it fell within the borders of a high compliance fully protected Marine Protected Area (MPA); ii) restricted - whether there were active restrictions on gears (e.g. bans on the use of nets, spearguns, or traps) or fishing effort (which could have included areas under customary tenure, where ‘outsiders’ were effectively excluded, as well as inside marine parks that were not necessarily no take); or iii) openly fished - regularly fished without effective restrictions. To determine these classifications, we used the expert opinion of the data providers, and triangulated this with a global database of MPA boundaries (32). As a sensitivity analysis, we also conducted analyses with a subset of fully protected MPAs that were $>2\text{km}^2$ and that have been protected for more than 4 years (see analysis section below).

2. Local Population Growth: We created a 100 km buffer around each reef cluster and used this to calculate human population within the buffer in 2000 and 2010 based on the Socioeconomic Data and Application Centre (SEDAC) gridded population of the world database (33). Population

growth was the proportional difference between the population in 2000 and 2010. We chose a 100km buffer as a reasonable range at which many key human impacts from population (e.g., land-use and nutrients) might affect reefs (34).

3. *Gravity*: We adapted the economic geography concept of *gravity* (13, 15, 35, 36) to examine the amount of human pressure within the surrounding 500km of a reef. Based on an analogy from Newtonian gravity ($\text{gravity} = \text{mass} / \text{distance}^2$), the gravity model (also called interactance) has been used by economists and geographers since the 1880s to measure the ‘gravitational pull’ of a wide range of economic interactions such as trade and migration flows (13). This application of the gravity concept infers that potential economic interactions increase with human population size (e.g. mass), but decay exponentially with the effective distance between two points. To calculate gravity, we gathered data on both population estimates and a surrogate for distance: travel time.

Population estimations

We gathered population estimates for every 1-by-1 km populated cell within a 500km radius of each reef site using the LandScan™ 2011 database (37). We chose a 500km radius from the nearest settlement as the maximum distance any non-market fishing activities for fresh reef fish are likely to occur.

Travel time calculation

For each populated cell within 500km, we then used a cost-distance algorithm that computes the least 'cost' (in minutes) of travelling to the reef site. Cost was based on a raster grid of land cover, road networks, and shorelines data and estimated travel time over different surfaces (38).

Gravity computation

We first calculated a value for the "gravitational pull" exerted by each populated cell within 500km of a reef site, by dividing the population of that cell by the squared travel time to the reef site. We then summed the gravity values for all cells within 500km of each reef site to measure the total gravity of human pressure that a given reef is experiencing. This application of the gravity concept infers that potential interactions increase with population size, but decay non-linearly with the effective distance. Although different exponents can be used, we used the traditional application of dividing by squared distance (in our case travel time)(13). This application emphasizes a non-linear decay in the propensity for interactions as distance from people to the reef increases. Our rationale for calculating gravity using squared travel time in the denominator (as opposed to just travel time) is based on the idea that our reef site is likely only one of multiple reefs that could potentially be harvested, and that the number of potential alternative reefs that could be harvested should increase with the area covered by a radius from any populated cell (i.e., based on area not linear distance). Since the decision to fish on a given reef is likely dependent on how that reef compares with all other alternatives, it makes sense that fishing pressure at any reef site will also decline by distance squared (i.e.

comparing with all other reefs within a similar distance) rather than linear distance (i.e. comparing only with other reefs along the same path). To test whether this rationale to use squared travel time is supported by our data, we developed gravity metrics using a range of exponents (1 , 2 , 3) and used leave-one-out cross-validation for model selection to determine the best fit. Squared travel time performed best for all three response variables, which supports our decision to use that for our analysis. However, for parrotfish scraping, its travel time (i.e. exponent 1) was within the standard error. Due to the potential ambiguity in the parrotfish scraping potential, we ran a sensitivity test, calculating how the probabilities of achieving goals change along a gradient of human pressure using a gravity metric calculated using the first exponent (i.e. travel time in the denominator). There were no discernible differences between our results, suggesting that our decision to use travel time squared as opposed to travel time in the denominator did not meaningfully impact our results.

4. Human Development Index (HDI): HDI is a summary measure of human development encompassing: life expectancy, education, and per capita income. We obtained the HDI measure from the United Nations Development Program for 2010. In cases where HDI values were not available specific to the State (e.g. Hawaii), we used the national (e.g. USA) HDI value, and in other cases (e.g. Marshall Islands) we had to calculate HDI from life expectancy, education, and per capita income statistics.

5. Population size. For each nation/state, we determined the size of the human

Population in 2010. Data were derived mainly from the national census reports CIA fact book (<https://www.cia.gov/library/publications/the-world-factbook/rankorder/2119rank.html>), and Wikipedia (https://en.wikipedia.org/wiki/Main_Page).

6. *National Reef Fish Landings*: Reconstructed reef fish catch estimates (in metric tonnes) were obtained from the Sea Around Us Project (SAUP) catch database (<http://www.searoundus.org>)(39). We used estimates corresponding to 2010 and only included reef associated species. We calculated the catch per unit area (catch/km²/y) by dividing a nation/state's catch by the its estimated reef area (40).

7. *Oceanic productivity*: We examined oceanic net productivity for each reef following the procedure described by (41). We delimited a 100 km buffer around each of our reef clusters, we removed shallow waters pixels (those that intersected or were contained within the depth contour of 30m from the General Bathymetric Chart of the Oceans 2014 (<http://www.gebco.net/>), a global gridded bathymetry dataset) and then calculated the average of monthly chlorophyll-a concentration (proxy for phytoplankton biomass) using data provided at a 4km-resolution by Aqua MODIS (Moderate Resolution Imaging Spectro-radiometer) for years 2005 to 2010.

8. *Climate stress*. We included an index of climate stress for corals, developed by (42), which incorporated 11 different environmental conditions, including the mean and variability of sea-surface temperature, tidal range, ultraviolet radiation, a doldrum index, and chlorophyll.

Analyses

We first looked for collinearity among our covariates using bivariate correlations and variance inflation factor estimates. This led to the exclusion of several covariates (not described above): i) *Gross Domestic Product* (purchasing power parity); ii) *Rule of Law* (World Bank governance index); iii) *Control of Corruption* (World Bank governance index (43)); iv) *Sedimentation*; v) *Tourism* (tourist arrivals from the World Tourism Organization's Compendium of Tourism Statistics relative to land area); vi) *Atoll* (i.e., a binary metric of whether the reef site was on an atoll or not); vii) *Frequency of storms since 1980* (<http://weather.unisys.com/hurricane>); viii) *Environmental performance index (EPI)* (<https://epi.envirocenter.yale.edu/>). ; and ix) the *GINI index* (measure of a nation/state's inequality). Although the GINI index was not strongly correlated with other covariates, there were numerous missing values, so that potential covariate was removed. All other covariates had Variance Inflation Factor scores less than 2 (indicating multicollinearity was not a concern). Care must be taken in causal attribution of covariates that were significant in our models, but demonstrated collinearity with candidate covariates that were removed during the aforementioned process. Critically, our metric of total gravity was colinear with atoll (i.e., most remote or low gravity reefs are atolls) but when we restricted the analyses to only non-atolls the results did not change. Additionally, correlations between mean body size of the fish assemblage (length, cm) and our response variables: biomass ($r=0.73$), parrotfish scraping potential ($r=0.2$), and trait diversity ($r=0.4$) suggest that mean body size is only predictive of biomass.

Multilevel models

To quantify the multi-scale social, environmental, and economic factors affecting the three ecological metrics, we modelled each response variable separately using multilevel models that explicitly recognized the three scales of spatial organization: reef site, reef cluster and nation/state. Models were run using a Bayesian approach using the Hamiltonian Monte Carlo algorithm implemented in Stan through the brms package (44) for 10000 iterations, and a 9000 burn in. This left 4000 samples in the posterior distribution of each parameter (four chains). We did not have a priori information about parameter distributions; thus, the posterior estimates were informed by the data alone (i.e. weakly informative priors). Convergence was monitored by running four chains from different starting points, examining posterior chains and distribution for stability, and checking that the potential scale reduction factor (also termed $R_{\hat{}}$) was close to 1. We employed a gaussian distribution to analyze biomass of reef fish above 20 cm (log +1 transformed) and trait diversity (log transformed), and used a hurdle-lognormal to analyze parrotfish scraping potential because the data for this metric contained a large number of zeros (31 %). The hurdle model is a two-part model composed of (i) a binomial distribution and a logit link function to predict the probability of observing the herbivory function (i.e., whether the response outcome is positive or zero) and (ii) a lognormal distribution for the non-zero data.

For each model, we set reef cluster and nation/state as random effects to account for the hierarchical nature of the data (i.e. reef sites nested in reef clusters, reef clusters nested in nations/states). For each metric, we tested two alternate models: a null model, consisting only of the hierarchical units of observation (that is, intercepts-only) and a full model that included all

of our covariates (potential drivers) of interest. We used the null model as a baseline against which we could ensure through leave-one-out cross-validation information criteria (LOOIC) (45) that our full model performed better than a model with no covariate information. To account for any methodological effects, sampling area, census method, sampled habitat and depth were also included in all the models as covariates. To control for sampling effects, we marginalized response variables by subtracting the estimated sampling standardized mean model effects to the observed response variables. For all the analyses, continuous covariates were standardized (mean centered and divided by 2 standard deviations). To examine model fit and homoscedasticity, we conducted posterior predictive checks, checked residuals against fitted values and ensured residuals followed expected distributions around zero (e.g., for the gaussian distribution models we checked that residuals were normally distributed around zero). We also checked the residuals against all covariates included in the models, and the covariates described above that were not included in the models (primarily due to collinearity). The residuals of each of the three models showed no patterns with these covariates, suggesting they would not explain additional information in our models. Additionally, to account for the potential effect that MPA size and age could have on our response variables we ran two different analyses: (i) where we included all the high compliance MPAs in our data irrespective of size and age (N=106 reef sites); and (ii) where we only retained MPA sites that were above a minimum threshold of at least 2 km² and older than 4 years, (N=61 reef sites). These inclusion criteria were informed by the literature on MPA effectiveness, which suggests that a diameter of 1-2km (1-3km²) is required to achieve partial protection (46), but were also constrained by our sample; a more conservative cutoff of say 10km² and 10 years would have left only 16 reef sites. In the main manuscript, we report (i),

but highlight the differences between (i) and (ii) in Fig. S7. All analyses were undertaken using R (3.02) statistic package.

Reference conditions and targets

We defined reference conditions for each ecological metric using the 0.9 quantile of the marginalized response variables accounting for sampling, habitat sampled, and sampling location (i.e., response variables minus the random effects and the model estimated effect sizes of depth category, reef habitat and sampling method). Thus, reference conditions are for average sampling area and “Slopes”, “4-10m” and “Standard belt transects”. As expected, the 90% reference point values for the fisheries target (biomass above 20 cm) was slightly below the expected total biomass in remote locations (47). Consequently, we then set targets of 25, 50, and 75% of these reference point conditions, the lower two of which correspond to typical standing biomass levels of multispecies maximum sustainable yields (hypothesized to be between 25-50% of unfished biomass estimates (10, 48)). Meanwhile 75% of reference conditions is considered a more stringent conservation target. For consistency, we used the same reference conditions and targets (i.e. 25, 50, and 75% of reference conditions) for parrotfish scraping potential and trait diversity, although established ecological significance of these figures remains untested, and establishing benchmarks for these is an important area of future research, as is developing region-specific reference conditions. To avoid being overly prescriptive, we also ran our analyses for a range of reference conditions, based on 0.8 and 0.95 of the response variables, and incorporated the results in the supplemental information (Fig. S3-S4).

To estimate the probability of passing different thresholds under a gradient of gravity (e.g., Fig. 2), for each response variable, we simulated new data from the model posteriors where only gravity was modified (i.e., maintaining all the other covariates at average conditions, for slopes, 4-10 m of depth and standard belt transects and not including the random effects) and estimated the probability of the posterior samples being above or below the targets. To determine the probability of all three response variables passing the targets (i.e., co-occurrence of metrics), we used the subset of 1662 reef sites that had all three ecological metrics and multiplied the probabilities (i.e., assuming independence).

Potential conservation gains from management for our reef sites

To estimate the number of fished sites that would pass different targets if management (i.e., high compliance marine MPAs or restrictions) were implemented, we simulated new data for the posterior distributions maintaining sampling consistent (i.e., sampling method and sampling area) but allowing individual sites to have their own socio-ecological context (e.g., habitat, depth, HDI, random effects). Then, we changed their protection (from openly fished to high compliance MPAs or restricted) and simulated a new set of data based on that condition. This allowed us to estimate the number of our sites that could potentially pass different thresholds if management was implemented given the effect of management in our model and a site's own environmental and socio-economic context. We report the high compliance MPA results in the main manuscript and the restricted fishing in the supplemental information.

Fig. S1.

Correlations between the three key ecological metrics supported by fish communities on coral reefs.

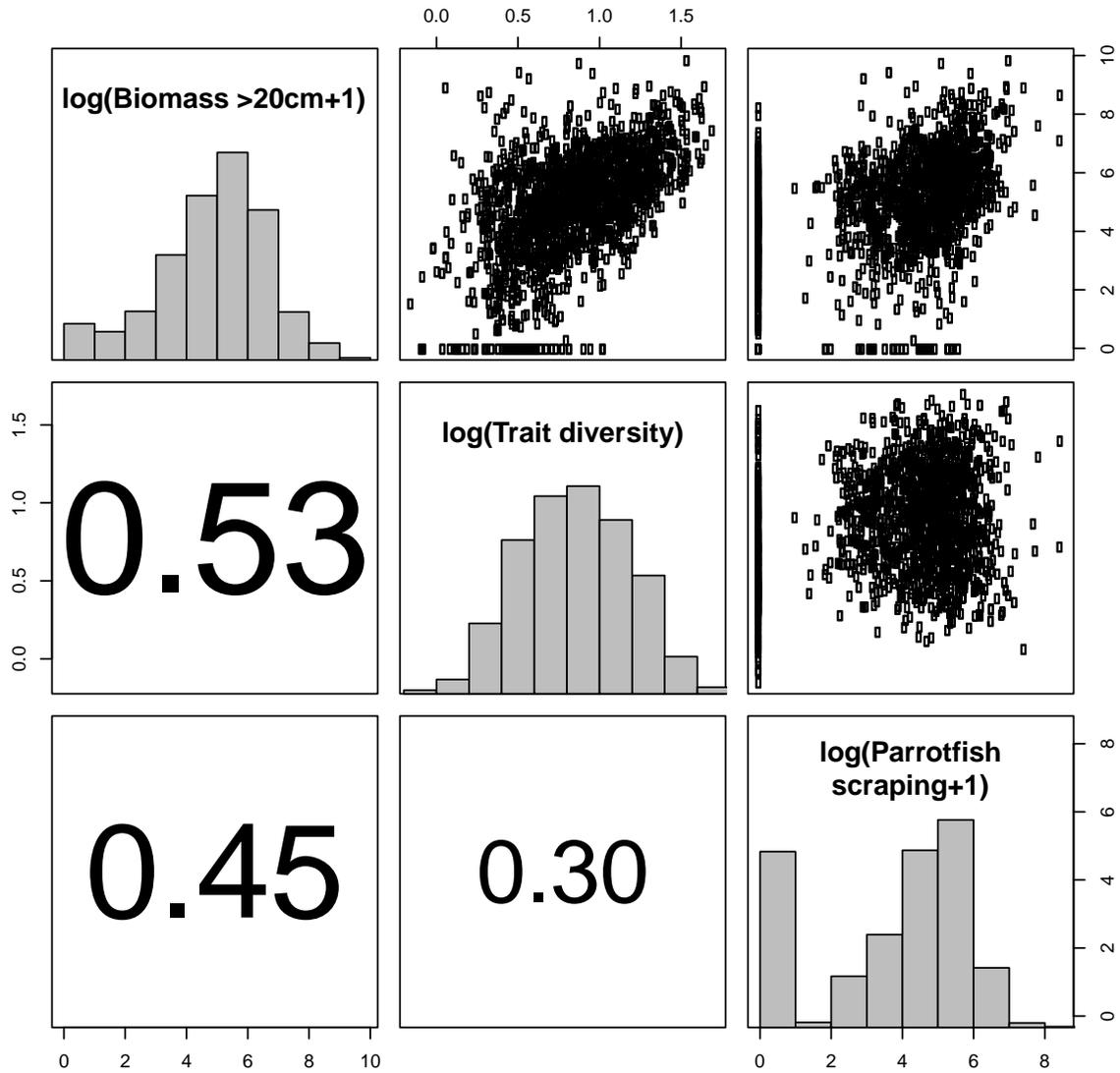


Fig. S2.

Effect size of eight socioeconomic drivers, management, sampling, and environmental conditions on three fish metrics. (A) biomass of reef fish >20cm. (B) parrotfish scraping potential. (C) trait diversity. Total gravity was the most consistent socioeconomic covariate, demonstrating strong negative relationships with fish biomass and trait diversity, and a weaker negative relationship with parrotfish scraping potential (posterior slope had 65.4% of the samples negative). Continuous covariates were standardized (mean centered and divided by 2 standard deviations), while response variables were not. Thus, effect sizes are standardized within columns only. Parameter estimates are Bayesian posterior mean values and 95% uncertainty intervals (UI). Red or green dots indicate negative or positive relationships, respectively, where the 95% UI does not overlap 0. A Hurdle model was used for parrotfish scraping (b).

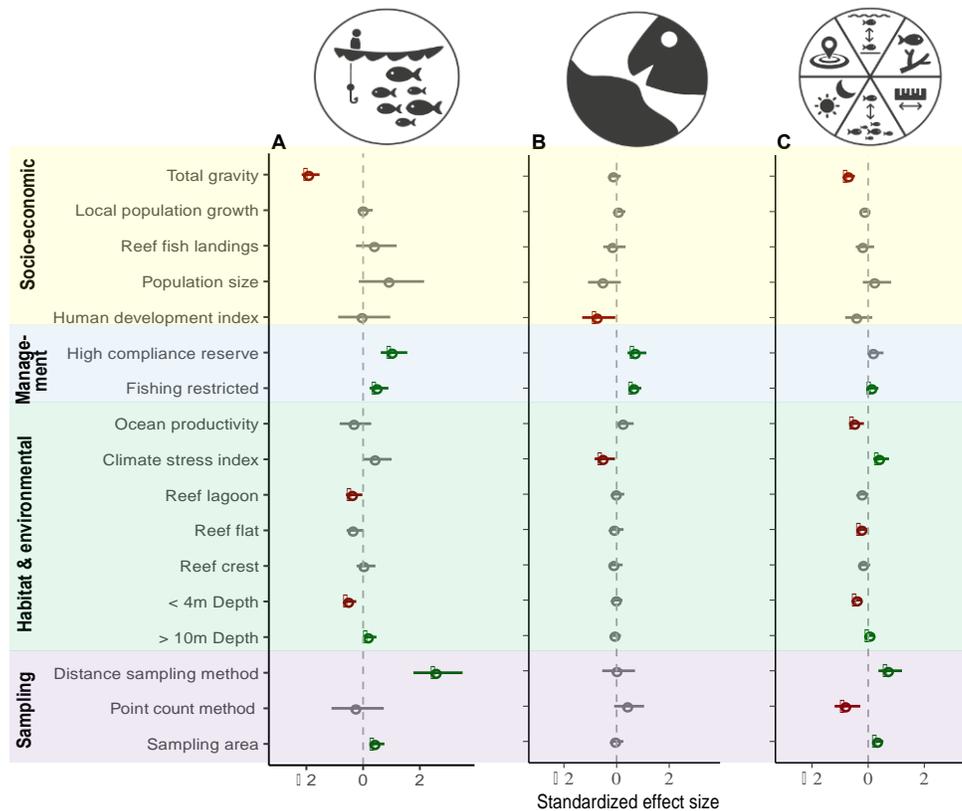


Fig. S3.

The estimated probability of openly fished reef sites having 25, 50, and 75% of reference conditions (light, medium, and dark purple, respectively). (A) a combination of fish biomass (>20cm), parrotfish scraping potential, trait diversity, and (B-D) each metric, respectively, along a gradient of human pressure (gravity). Separate estimates are provided for reef sites in fully protected MPAs (E-H) and with restricted fishing (I-L). To highlight how the potential benefits of management change along a gradient of human pressure (gravity), we extracted the difference in the probability of achieving each target between MPAs and openly fished sites (M-P), restricted and openly fished areas (Q-T), and MPAs and restricted areas (U-X). We plotted the partial effect of the relationship between gravity and each benchmark by setting all other continuous covariates to 0 (because they were all standardized) and all categorical covariates to their most common category (i.e. 4-10m for depth, slope for habitat, standard belt transect for census method).

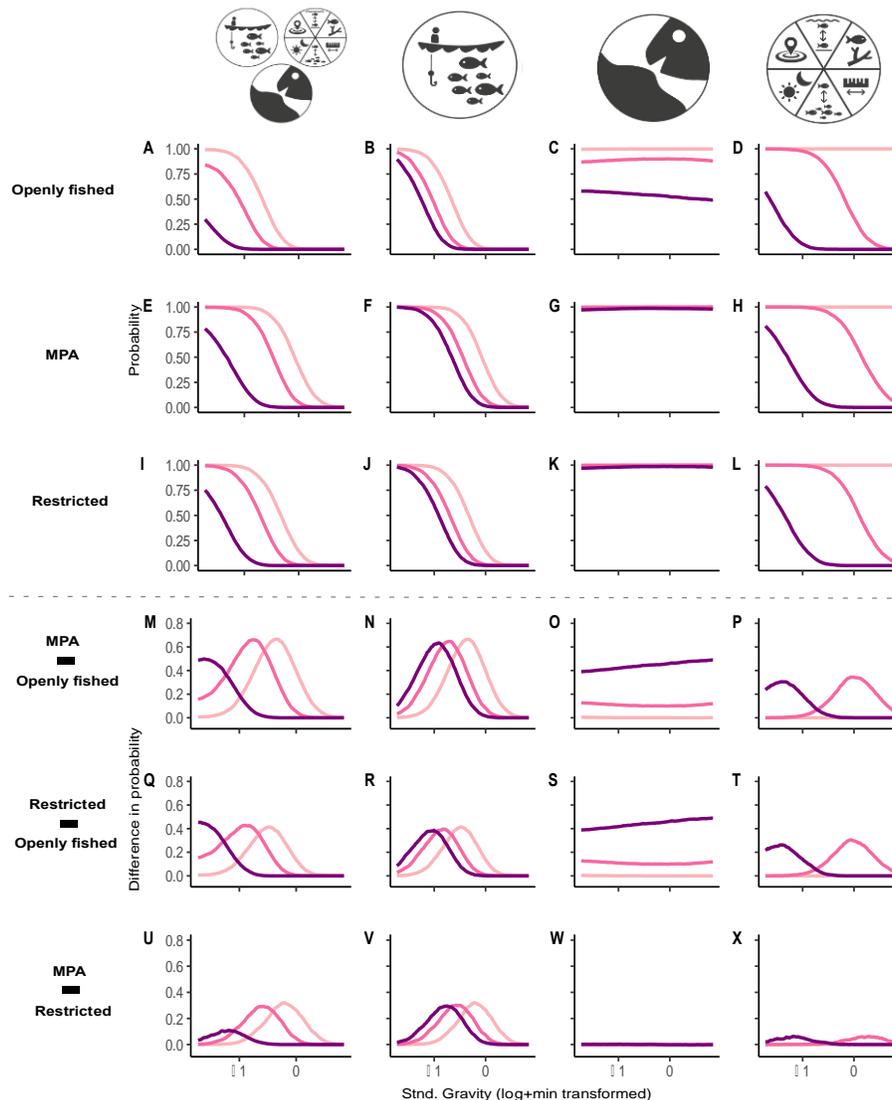


Fig. S4.

The estimated probability of openly fished reef sites having 25, 50, and 75% of reference conditions (light, medium, and dark purple, respectively). (A) a combination of fish biomass (>20cm), parrotfish scraping potential, trait diversity, and (B-D) each metric, respectively, along a gradient of human pressure (gravity). Separate estimates are provided for reef sites in MPAs (E-H) and with restricted fishing (I-L). To highlight how the potential benefits of management change along a gradient of human pressure (gravity), we extracted the difference in the probability of achieving each target between MPAs and openly fished sites (M-P), restricted and openly fished areas (Q-T), and MPAs and restricted areas (U-X). We plotted the partial effect of the relationship between gravity and each benchmark by setting all other continuous covariates to 0 (because they were all standardized) and all categorical covariates to their most common category (i.e. 4-10m for depth, slope for habitat, standard belt transect for census method).

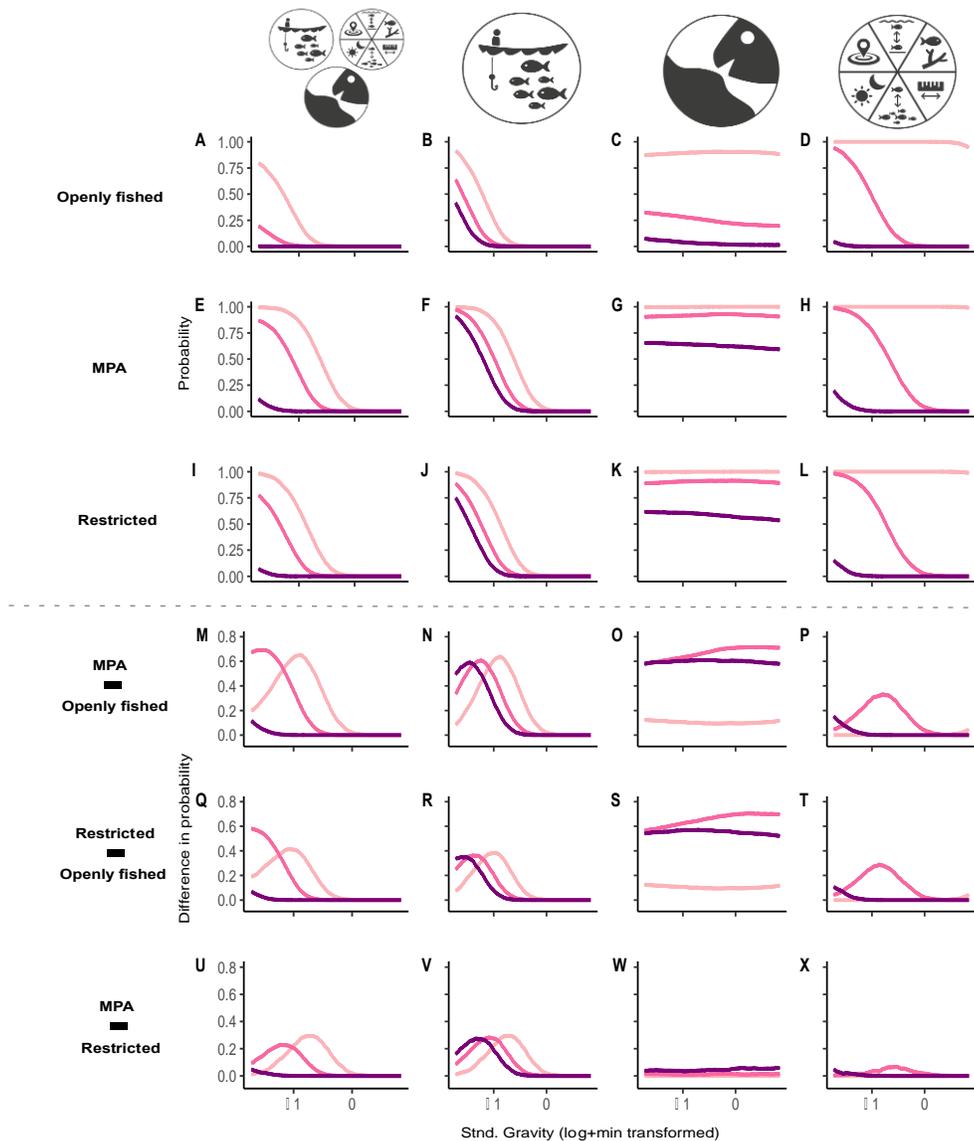


Fig. S5.

The scaled distribution of covariates for our sample of reefs (blue) and for all tropical reefs globally (grey). Our sampled reefs display a reasonably similar distribution and range for most covariates. Note that the global gravity values were only available rounded to the nearest integer, therefore to directly compare with our site level values, we used a log+1 transformation, rather than log+minimum transformation as used in the rest of the manuscript.

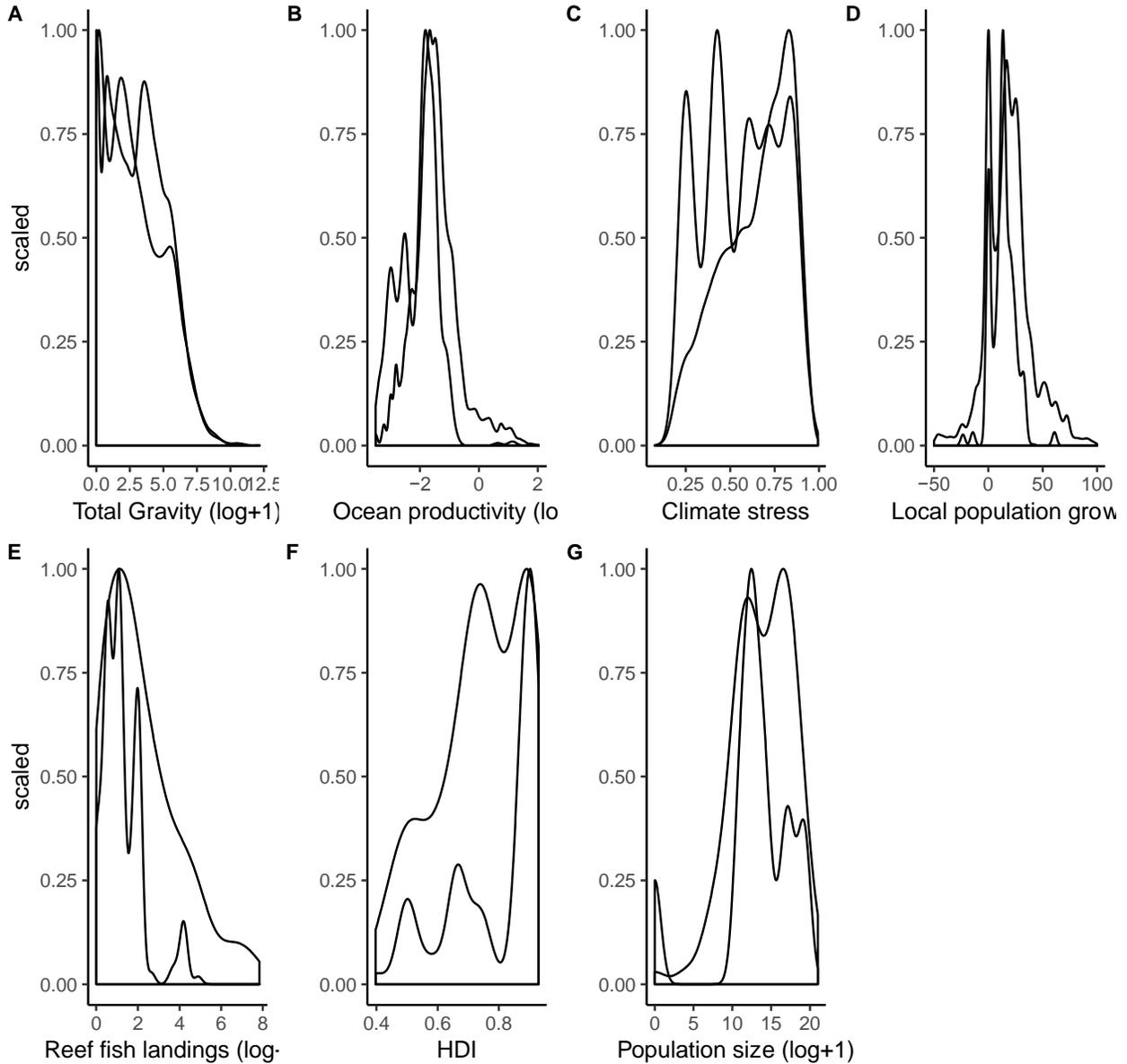


Fig. S6.

Conservation target outcomes from simulating the implementation of fishing restrictions in openly fished sites. Alluvial plots show the change in the number of sites expected to achieve key conservation targets if fisheries restrictions were implemented in our openly fished sites for (A) simultaneously meeting fish biomass, parrotfish scraping potential, and trait diversity, and (B-D) each goal, respectively.

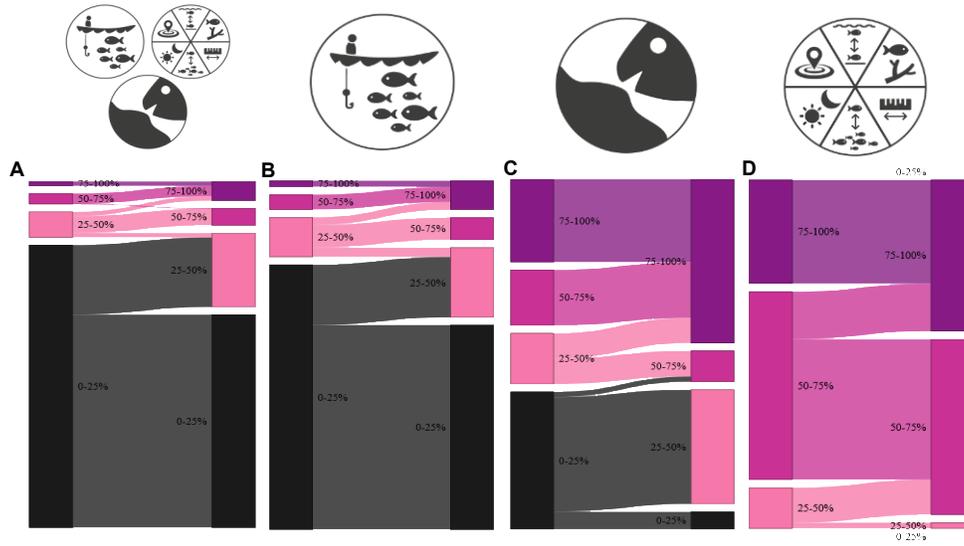


Fig. S7.

Difference in probability of achieving specific targets between the restricted subset of fully protected Marine Protected Areas (MPAs) (>2km² and 4 years old, n=61) and all MPAs in our sample (n=106). (A) simultaneously meeting fish biomass, parrotfish scraping potential, and trait diversity, and (B-D) each goal, respectively. Alluvial plots show the change in the number of sites expected to achieve key conservation targets if the marine reserves >2km² and 4 years old (based on our restricted subset) were implemented in our openly fished sites for (E) simultaneously meeting fish biomass, parrotfish scraping potential, and trait diversity, and (F-H) each goal, respectively. Black <25%, light pink =25-50%, dark pink=50-75%, and purple >75% of reference conditions.

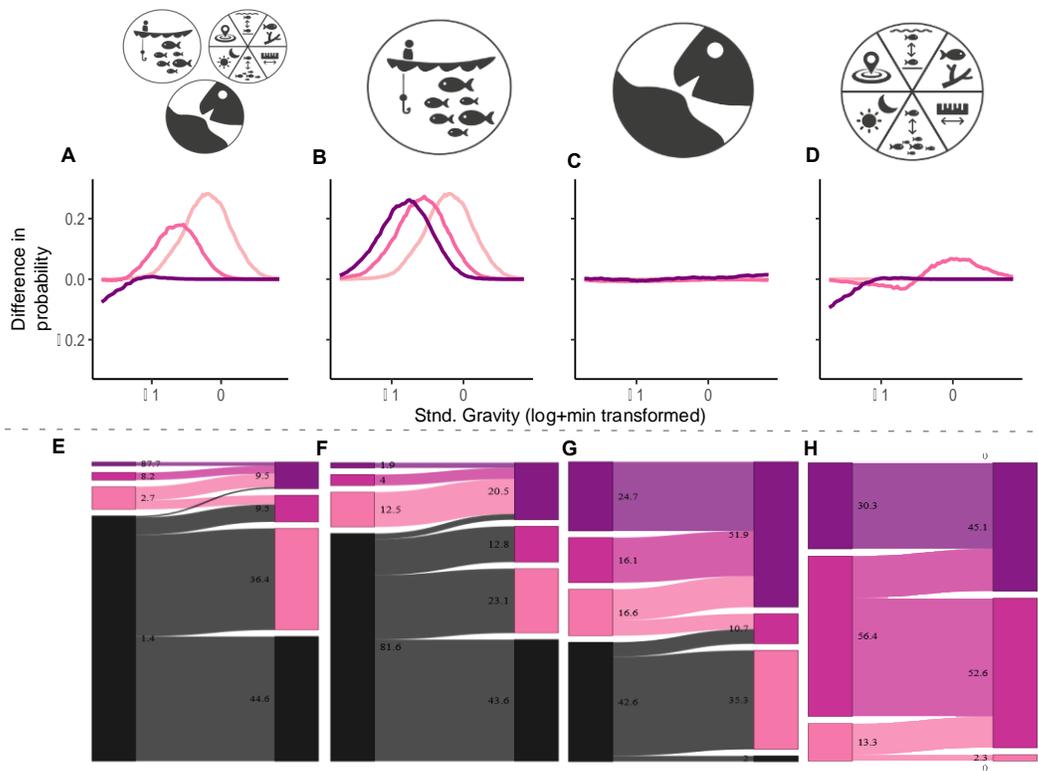


Table S1.

List of ‘Nation/states’ covered in study. In most cases, nation/state refers to an individual country, but can also include states (e.g. Hawaii), territories (e.g. British Indian Ocean Territory), or other jurisdictions.

Nation/States
American Samoa
Australia
Belize
Brazil
British Indian Ocean Territory
Cayman Islands
Colombia
Commonwealth of the Northern Mariana Islands
Comoro Islands
Cuba
Egypt
Federated States of Micronesia
Fiji
French Polynesia
Guam
Hawaii
Indonesia
Jamaica
Kenya
Kiribati
Madagascar
Maldives
Marshall Islands
Mauritius
Mayotte
Mexico
Mozambique
Netherlands Antilles
New Caledonia
Oman

Palau
Panama
Papua New Guinea
Philippines
PRIA
Reunion
Seychelles
Solomon Islands
Tanzania
Tonga
Venezuela

Table S2.**Justification of ecological metrics**

Biomass of fish above 20 cm	Large fish are both key to sustain ecosystem functioning and common fishery targets. We selected a 20 cm cut-off point because it includes large fish and “plate-sized” fish, targeting not only the most valuable fish but also the fish destined to food consumption (49). Additionally, large fish exert top-down control on ecosystems, regulating the structure and functions of reef ecosystems (50). Biomass captures both the size and number of fish above 20 cm in the system, which dictates the magnitude of the function (51). Biomass of fish above 20 cm is expected to decline rapidly as human impacts intensify (10), and there is empirical evidence that management can allow the recovery of large species (52).
Parrotfish scraping	Herbivory mediates the competition between corals and algae. Bioerosion removes dead reef structures, providing suitable substrate for coral recruitment. Parrotfish are among the most important groups of herbivorous fish on coral reefs performing processes of algae removal and contributing to bioerosion, hence maintenance of good condition for reef growth. Herbivory is expected to decline as human impacts intensify (49) and respond positively to management (53).
Trait diversity	The diversity of ecological traits supported by species can represent the range of potential ecological roles present in a given community (54, 55). A broader range of traits are assumed to provide a greater contribution to key ecosystem processes (e.g. biomass production, nutrient cycling) and cultural services (e.g. aesthetic value) than a smaller range of traits (55–57). We estimated trait diversity (TD) using the Chao’s $FD_{q=1}$ index which is a generalization of the taxonomic Shannon’s entropy index (7). This index is high when both the dissimilarity of species’ traits (e.g. diet, size) and the spread of biomass across these traits are high. We posit that TD should generally decrease as human impacts increase, because activities such as fisheries selectively target species with specific traits, which can reduce the trait space occupied and the balance of biomass among traits, and thus TD (58, 59).

Table S3.

Summary of social and environmental covariates. Further details can be found in Methods. The smallest scale is the individual reef site. Reef clusters consist of clusters of reef sites within 4km of each other. Nation/states generally correspond to country, but can also include territories or states, particularly when geographically isolated (e.g. Hawaii).

Covariate	Description	Scale	Key data sources
Local population growth	Difference in local human population (i.e. 100km buffer around our reef clusters) between 2000-2010	Reef cluster	Socioeconomic Data and Application Centre (SEDAC) gridded population of the world database (33)
'Gravity' of human pressure	For each populated cell within a 500km radius of a reef site, we divided the population of that cell by the squared travel time between the reef site and the cell to get a gravity value (i.e. how much "gravitational pull" that population was exerting on the reef site). This was then summed for all cells to	Reef site	Human population size, land cover, road networks, coastlines

get the total gravity of human pressure.

Management status	Whether the reef site is openly fished, restricted (e.g. effective gear bans or effort restrictions), or unfished	Reef site	Expert opinion, global map of marine protected areas.
Human Development index	A summary measure of human development encompassing: a long and healthy life, being knowledgeable and have a decent standard of living. We used linear and quadratic functions for HDI.	Nation/state	United Nations Development Programme
Population Size	Total population size of the jurisdiction	Nation/ state	World Bank, census estimates, Wikipedia
Fish landings	Landings of reef fish (tons) per Km ² of reef	Nation/ state	Sea Around Us Project (39)
Climate stress	A composite metric comprised of 11 different environmental	Reef cluster	Maina et al. (42)

	variables that are related to coral mortality from bleaching		
Productivity	The monthly average (2005-2010) oceanic productivity	Reef cluster	Gove et al. 2013 (41), Aqua MODIS
Habitat	Whether the reef site is a slope, crest, flat, or back reef/lagoon	Reef site	Primary data
Depth	Depth of the ecological survey (<4m, 4.1-10m, >10m)	Reef site	Primary data
Sampling technique	Whether the data collector used point count, line transects, or distance sampling	Reef site	Primary data
Area Sampled	The size of the area sampled by the data provider (in m ²)	Reef site	Primary data

Table S4.

List of fish families included in this study for both the trait diversity and the biomass above 20 cm response variables.

Fish family	Common family name
Acanthuridae	Surgeonfishes
Balistidae	Triggerfishes
Carangidae	Jacks
Diodontidae	Porcupinefishes
Ephippidae	Batfishes
Haemulidae	Sweetlips
Kyphosidae	Drummers
Labridae	Wrasses
Lethrinidae	Emperors
Lutjanidae	Snappers
Monacanthidae	Filefishes
Mullidae	Goatfishes
Nemipteridae	Coral Breams
Pinguipedidae	Sandperches
Pomacanthidae	Angelfishes
Scaridae	Wrasses and Parrotfish
Serranidae	Groupers
Siganidae	Rabbitfishes
Sparidae	Porgies
Synodontidae	Lizardfishes
Tetraodontidae	Pufferfishes
Zanclidae	Moorish Idol

References and Notes

1. T. P. Hughes, M. L. Barnes, D. R. Bellwood, J. E. Cinner, G. S. Cumming, J. B. C. Jackson, J. Kleypas, I. A. van de Leemput, J. M. Lough, T. H. Morrison, S. R. Palumbi, E. H. van Nes, M. Scheffer, Coral reefs in the Anthropocene. *Nature* **546**, 82–90 (2017). [doi:10.1038/nature22901](https://doi.org/10.1038/nature22901) [Medline](#)
2. A. V. Norström, M. Nyström, J.-B. Jouffray, C. Folke, N. A. J. Graham, F. Moberg, P. Olsson, G. J. Williams, Guiding coral reef futures in the Anthropocene. *Front. Ecol. Environ.* **14**, 490–498 (2016). [doi:10.1002/fee.1427](https://doi.org/10.1002/fee.1427)
3. E. K. Pikitch, C. Santora, E. A. Babcock, A. Bakun, R. Bonfil, D. O. Conover, P. Dayton, P. Doukakis, D. Fluharty, B. Heneman, E. D. Houde, J. Link, P. A. Livingston, M. Mangel, M. K. McAllister, J. Pope, K. J. Sainsbury, Ecosystem-based fishery management. *Science* **305**, 346–347 (2004). [doi:10.1126/science.1098222](https://doi.org/10.1126/science.1098222) [Medline](#)
4. Materials and methods are available as supplementary materials.
5. Y.-M. Bozec, S. O’Farrell, J. H. Bruggemann, B. E. Luckhurst, P. J. Mumby, Tradeoffs between fisheries harvest and the resilience of coral reefs. *Proc. Natl. Acad. Sci. U.S.A.* **113**, 4536–4541 (2016). [doi:10.1073/pnas.1601529113](https://doi.org/10.1073/pnas.1601529113) [Medline](#)
6. J. E. Duffy, J. S. Lefcheck, R. D. Stuart-Smith, S. A. Navarrete, G. J. Edgar, Biodiversity enhances reef fish biomass and resistance to climate change. *Proc. Natl. Acad. Sci. U.S.A.* **113**, 6230–6235 (2016). [doi:10.1073/pnas.1524465113](https://doi.org/10.1073/pnas.1524465113) [Medline](#)
7. A. Chao, C.-H. Chiu, S. Villéger, I.-F. Sun, S. Thorn, Y.-C. Lin, J.-M. Chiang, W. B. Sherwin, An attribute-diversity approach to functional diversity, functional beta diversity, and related (dis)similarity measures. *Ecol. Monogr.* **89**, e01343 (2019). [doi:10.1002/ecm.1343](https://doi.org/10.1002/ecm.1343)
8. M. A. MacNeil, The politics, science and policy of reference points for resource management. *Environ. Conserv.* **40**, 297–301 (2013). [doi:10.1017/S0376892913000386](https://doi.org/10.1017/S0376892913000386)
9. G. M. Mace, R. Lande, Assessing extinction threats: Toward a reevaluation of IUCN threatened species categories. *Conserv. Biol.* **5**, 148–157 (1991). [doi:10.1111/j.1523-1739.1991.tb00119.x](https://doi.org/10.1111/j.1523-1739.1991.tb00119.x)
10. T. R. McClanahan, N. A. J. Graham, M. A. MacNeil, N. A. Muthiga, J. E. Cinner, J. H. Bruggemann, S. K. Wilson, Critical thresholds and tangible targets for ecosystem-based management of coral reef fisheries. *Proc. Natl. Acad. Sci. U.S.A.* **108**, 17230–17233 (2011). [doi:10.1073/pnas.1106861108](https://doi.org/10.1073/pnas.1106861108) [Medline](#)
11. H. J. Geist, E. F. Lambin, Proximate causes and underlying driving forces of tropical deforestation. *Bioscience* **52**, 143–150 (2002). [doi:10.1641/0006-3568\(2002\)052\[0143:PCAUDF\]2.0.CO;2](https://doi.org/10.1641/0006-3568(2002)052[0143:PCAUDF]2.0.CO;2)
12. J. E. Cinner, C. Huchery, M. A. MacNeil, N. A. J. Graham, T. R. McClanahan, J. Maina, E. Maire, J. N. Kittinger, C. C. Hicks, C. Mora, E. H. Allison, S. D’Agata, A. Hoey, D. A. Feary, L. Crowder, I. D. Williams, M. Kulbicki, L. Vigliola, L. Wantiez, G. Edgar, R. D. Stuart-Smith, S. A. Sandin, A. L. Green, M. J. Hardt, M. Beger, A. Friedlander, S. J. Campbell, K. E. Holmes, S. K. Wilson, E. Brokovich, A. J. Brooks, J. J. Cruz-Motta, D. J. Booth, P. Chabanet, C. Gough, M. Tupper, S. C. A. Ferse, U. R. Sumaila, D. Mouillot,

- Bright spots among the world's coral reefs. *Nature* **535**, 416–419 (2016).
[doi:10.1038/nature18607](https://doi.org/10.1038/nature18607) [Medline](#)
13. J. E. Anderson, The gravity model. *Annu. Rev. Econ.* **3**, 133–160 (2011).
[doi:10.1146/annurev-economics-111809-125114](https://doi.org/10.1146/annurev-economics-111809-125114)
 14. R. S. Steneck, P. J. Mumby, C. MacDonald, D. B. Rasher, G. Stoyle, Attenuating effects of ecosystem management on coral reefs. *Sci. Adv.* **4**, eaao5493 (2018).
[doi:10.1126/sciadv.aao5493](https://doi.org/10.1126/sciadv.aao5493) [Medline](#)
 15. J. E. Cinner, E. Maire, C. Huchery, M. A. MacNeil, N. A. J. Graham, C. Mora, T. R. McClanahan, M. L. Barnes, J. N. Kittinger, C. C. Hicks, S. D'Agata, A. S. Hoey, G. G. Gurney, D. A. Feary, I. D. Williams, M. Kulbicki, L. Vigliola, L. Wantiez, G. J. Edgar, R. D. Stuart-Smith, S. A. Sandin, A. Green, M. J. Hardt, M. Beger, A. M. Friedlander, S. K. Wilson, E. Brokovich, A. J. Brooks, J. J. Cruz-Motta, D. J. Booth, P. Chabanet, C. Gough, M. Tupper, S. C. A. Ferse, U. R. Sumaila, S. Pardede, D. Mouillot, Gravity of human impacts mediates coral reef conservation gains. *Proc. Natl. Acad. Sci. U.S.A.* **115**, E6116–E6125 (2018). [doi:10.1073/pnas.1708001115](https://doi.org/10.1073/pnas.1708001115) [Medline](#)
 16. P. J. Ferraro, S. K. Pattanayak, Money for nothing? A call for empirical evaluation of biodiversity conservation investments. *PLOS Biol.* **4**, e105 (2006).
[doi:10.1371/journal.pbio.0040105](https://doi.org/10.1371/journal.pbio.0040105) [Medline](#)
 17. J. P. W. Robinson, S. K. Wilson, J. Robinson, C. Gerry, J. Lucas, C. Assan, R. Govinden, S. Jennings, N. A. J. Graham, Productive instability of coral reef fisheries after climate-driven regime shifts. *Nat. Ecol. Evol.* **3**, 183–190 (2019). [doi:10.1038/s41559-018-0715-z](https://doi.org/10.1038/s41559-018-0715-z) [Medline](#)
 18. R. Hilborn, K. Stokes, J.-J. Maguire, T. Smith, L. W. Botsford, M. Mangel, J. Orensanz, A. Parma, J. Rice, J. Bell, K. L. Cochrane, S. Garcia, S. J. Hall, G. P. Kirkwood, K. Sainsbury, G. Stefansson, C. Walters, When can marine reserves improve fisheries management? *Ocean Coast. Manage.* **47**, 197–205 (2004).
[doi:10.1016/j.ocecoaman.2004.04.001](https://doi.org/10.1016/j.ocecoaman.2004.04.001)
 19. B. J. Bergseth, M. Roscher, Discerning the culture of compliance through recreational fisher's perceptions of poaching. *Mar. Policy* **89**, 132–141 (2018).
[doi:10.1016/j.marpol.2017.12.022](https://doi.org/10.1016/j.marpol.2017.12.022)
 20. J. Cinner, How behavioral science can help conservation. *Science* **362**, 889–890 (2018).
[doi:10.1126/science.aau6028](https://doi.org/10.1126/science.aau6028) [Medline](#)
 21. G. Post, J. Geldmann, Exceptional responders in conservation. *Conserv. Biol.* **32**, 576–583 (2018). [doi:10.1111/cobi.13006](https://doi.org/10.1111/cobi.13006) [Medline](#)
 22. J. Cinner, Data for: Meeting fisheries, ecosystem function, and biodiversity goals in a human-dominated world. James Cook University (2020);
<https://doi.org/10.25903/5e61a7f08ae39>.
 23. J. Zamborain-Mason. Code for: Meeting fisheries, ecosystem function, and biodiversity goals in a human-dominated world. Zenodo (2020); <https://zenodo.org/record/3697928#.Xnu-VdLtwk>.
 24. T. M. Daw, J. Maina, J. Cinner, J. Robinson, A. Wamukota, *The Spatial Behaviour of*

- Artisanal Fishers: Implications for Fisheries Management and Development (Fishers in Space), Final Report, December 2011* (2011); <https://www.researchgate.net/publication/321796381> The spatial behaviour of artisanal fishers Implications for fisheries management and development Fishers in Space Final Report.
25. R. Froese, D. Pauly, FishBase (2015); <http://www.fishbase.org>.
 26. A. S. Hoey, D. R. Bellwood, Limited functional redundancy in a high diversity system: Single species dominates key ecological process on coral reefs. *Ecosystems* **12**, 1316–1328 (2009). [doi:10.1007/s10021-009-9291-z](https://doi.org/10.1007/s10021-009-9291-z)
 27. R. M. Bonaldo, A. S. Hoey, D. R. Bellwood, *Oceanography and Marine Biology* (CRC, 2014), vol. 52; <https://www.taylorfrancis.com/books/9781482220667>).
 28. R. M. Bonaldo, D. R. Bellwood, Size-dependent variation in the functional role of the parrotfish *Scarus rivulatus* on the Great Barrier Reef, Australia. *Mar. Ecol. Prog. Ser.* **360**, 237–244 (2008). [doi:10.3354/meps07413](https://doi.org/10.3354/meps07413)
 29. L. Ong, K. N. Holland, Bioerosion of coral reefs by two Hawaiian parrotfishes: Species, size differences and fishery implications. *Mar. Biol.* **157**, 1313–1323 (2010). [doi:10.1007/s00227-010-1411-y](https://doi.org/10.1007/s00227-010-1411-y)
 30. J. H. Bruggemann, A. M. van Kessel, J. M. van Rooij, A. M. Breeman, Bioerosion and sediment ingestion by the caribbean parrotfish *Scarus vetula* and *Sparisoma viride*: Implications of fish size, feeding mode and habitat use. *Mar. Ecol. Prog. Ser.* **134**, 59–71 (1996). [doi:10.3354/meps134059](https://doi.org/10.3354/meps134059)
 31. D. Mouillot, S. Villéger, V. Parravicini, M. Kulbicki, J. E. Arias-González, M. Bender, P. Chabanet, S. R. Floeter, A. Friedlander, L. Vigliola, D. R. Bellwood, Functional over-redundancy and high functional vulnerability in global fish faunas on tropical reefs. *Proc. Natl. Acad. Sci. U.S.A.* **111**, 13757–13762 (2014). [doi:10.1073/pnas.1317625111](https://doi.org/10.1073/pnas.1317625111) [Medline](#)
 32. C. Mora, S. Andréfouët, M. J. Costello, C. Kranenburg, A. Rollo, J. Veron, K. J. Gaston, R. A. Myers, Ecology. Coral reefs and the global network of Marine Protected Areas. *Science* **312**, 1750–1751 (2006). [doi:10.1126/science.1125295](https://doi.org/10.1126/science.1125295) [Medline](#)
 33. Center for International Earth Science Information Network, *Gridded Population of the World, Version 4 (GPWv4): Population Count, Revision 11* (CIESIN, 2015); <https://sedac.ciesin.columbia.edu/data/set/gpw-v4-population-count-rev11/metadata>.
 34. M. A. MacNeil, S. R. Connolly, “Multi-scale patterns and processes in reef fish abundance,” in *Ecology of Fishes on Coral Reefs*, C. Mora, Ed. (Cambridge Univ. Press, 2015), pp. 116–126.
 35. J. E. Anderson, A theoretical foundation for the gravity equation. *Am. Econ. Rev.* **69**, 106–116 (1979).
 36. E. G. Ravenstein, The laws of migration. *J. Stat. Soc. Lond.* **48**, 167 (1885). [doi:10.2307/2979181](https://doi.org/10.2307/2979181)
 37. E. A. Bright, A. N. Rose, M. L. Urban, LandScan (2012); <https://landscan.ornl.gov/>.
 38. E. Maire, J. Cinner, L. Velez, C. Huchery, C. Mora, S. Dagata, L. Vigliola, L. Wantiez, M.

- Kulbicki, D. Mouillot, How accessible are coral reefs to people? A global assessment based on travel time. *Ecol. Lett.* **19**, 351–360 (2016). [doi:10.1111/ele.12577](https://doi.org/10.1111/ele.12577) [Medline](#)
39. D. Pauly, D. Zeller, *Sea Around Us Concepts, Design and Data* (2015); <http://www.seaaroundus.org/>.
40. UNEP-WCMC, *Global Distribution of Warm-Water Coral Reefs* (2010); <https://data.unep-wcmc.org/datasets/1>.
41. J. M. Gove, G. J. Williams, M. A. McManus, S. F. Heron, S. A. Sandin, O. J. Vetter, D. G. Foley, Quantifying climatological ranges and anomalies for Pacific coral reef ecosystems. *PLOS ONE* **8**, e61974 (2013). [doi:10.1371/journal.pone.0061974](https://doi.org/10.1371/journal.pone.0061974) [Medline](#)
42. J. Maina, T. R. McClanahan, V. Venus, M. Ateweberhan, J. Madin, Global gradients of coral exposure to environmental stresses and implications for local management. *PLOS ONE* **6**, e23064 (2011). [doi:10.1371/journal.pone.0023064](https://doi.org/10.1371/journal.pone.0023064) [Medline](#)
43. World Bank, *Worldwide Governance Indicators* (2015); <http://info.worldbank.org/governance/wgi/>.
44. P. C. Bürkner, brms: An R package for Bayesian multilevel models using Stan. *J. Stat. Softw.* **80**, (2017). [doi:10.18637/jss.v080.i01](https://doi.org/10.18637/jss.v080.i01)
45. A. Vehtari, A. Gelman, J. Gabry, Practical Bayesian model evaluation using leave-one-out cross-validation and WAIC. *Stat. Comput.* **27**, 1433 (2017). [doi:10.1007/s11222-016-9709-3](https://doi.org/10.1007/s11222-016-9709-3)
46. N. C. Krueck, G. N. Ahmadi, A. Green, G. P. Jones, H. P. Possingham, C. Riginos, E. A. Treml, P. J. Mumby, Incorporating larval dispersal into MPA design for both conservation and fisheries. *Ecol. Appl.* **27**, 925–941 (2017). [doi:10.1002/eap.1495](https://doi.org/10.1002/eap.1495) [Medline](#)
47. T. R. McClanahan, R. E. Schroeder, A. M. Friedlander, L. Vigliola, L. Wantiez, J. E. Caselle, N. A. J. Graham, S. Wilson, G. J. Edgar, R. D. Stuart-Smith, R. M. Oddenyo, J. E. Cinner, Global baselines and benchmarks for fish biomass: Comparing remote reefs and fisheries closures. *Mar. Ecol. Prog. Ser.* **612**, 167–192 (2019). [doi:10.3354/meps12874](https://doi.org/10.3354/meps12874)
48. B. Worm, R. Hilborn, J. K. Baum, T. A. Branch, J. S. Collie, C. Costello, M. J. Fogarty, E. A. Fulton, J. A. Hutchings, S. Jennings, O. P. Jensen, H. K. Lotze, P. M. Mace, T. R. McClanahan, C. Minto, S. R. Palumbi, A. M. Parma, D. Ricard, A. A. Rosenberg, R. Watson, D. Zeller, Rebuilding global fisheries. *Science* **325**, 578–585 (2009). [doi:10.1126/science.1173146](https://doi.org/10.1126/science.1173146) [Medline](#)
49. S. M. W. Reddy, A. Wentz, O. Aburto-Oropeza, M. Maxey, S. Nagavarapu, H. M. Leslie, Evidence of market-driven size-selective fishing and the mediating effects of biological and institutional factors. *Ecol. Appl.* **23**, 726–741 (2013). [doi:10.1890/12-1196.1](https://doi.org/10.1890/12-1196.1) [Medline](#)
50. N. K. Dulvy, R. P. Freckleton, N. V. C. Polunin, Coral reef cascades and the indirect effects of predator removal by exploitation. *Ecol. Lett.* **7**, 410–416 (2004). [doi:10.1111/j.1461-0248.2004.00593.x](https://doi.org/10.1111/j.1461-0248.2004.00593.x)
51. D. R. Bellwood, A. S. Hoey, T. P. Hughes, Human activity selectively impacts the ecosystem roles of parrotfishes on coral reefs. *Proc. Biol. Sci.* **279**, 1621–1629 (2012).

[doi:10.1098/rspb.2011.1906](https://doi.org/10.1098/rspb.2011.1906) [Medline](#)

52. G. J. Edgar, R. D. Stuart-Smith, T. J. Willis, S. Kininmonth, S. C. Baker, S. Banks, N. S. Barrett, M. A. Becerro, A. T. F. Bernard, J. Berkhout, C. D. Buxton, S. J. Campbell, A. T. Cooper, M. Davey, S. C. Edgar, G. Försterra, D. E. Galván, A. J. Irigoyen, D. J. Kushner, R. Moura, P. E. Parnell, N. T. Shears, G. Soler, E. M. A. Strain, R. J. Thomson, Global conservation outcomes depend on marine protected areas with five key features. *Nature* **506**, 216–220 (2014). [doi:10.1038/nature13022](https://doi.org/10.1038/nature13022) [Medline](#)
53. T. P. Hughes, M. J. Rodrigues, D. R. Bellwood, D. Ceccarelli, O. Hoegh-Guldberg, L. McCook, N. Moltschanivskyj, M. S. Pratchett, R. S. Steneck, B. Willis, Phase shifts, herbivory, and the resilience of coral reefs to climate change. *Curr. Biol.* **17**, 360–365 (2007). [doi:10.1016/j.cub.2006.12.049](https://doi.org/10.1016/j.cub.2006.12.049) [Medline](#)
54. R. D. Stuart-Smith, A. E. Bates, J. S. Lefcheck, J. E. Duffy, S. C. Baker, R. J. Thomson, J. F. Stuart-Smith, N. A. Hill, S. J. Kininmonth, L. Airoidi, M. A. Becerro, S. J. Campbell, T. P. Dawson, S. A. Navarrete, G. A. Soler, E. M. A. Strain, T. J. Willis, G. J. Edgar, Integrating abundance and functional traits reveals new global hotspots of fish diversity. *Nature* **501**, 539–542 (2013). [doi:10.1038/nature12529](https://doi.org/10.1038/nature12529) [Medline](#)
55. G. D. Tilman, The influence of functional diversity and composition on ecosystem processes. *Science* **277**, 1300–1302 (1997). [doi:10.1126/science.277.5330.1300](https://doi.org/10.1126/science.277.5330.1300)
56. N. Gross, Y. L. Bagousse-Pinguet, P. Liancourt, M. Berdugo, N. J. Gotelli, F. T. Maestre, Functional trait diversity maximizes ecosystem multifunctionality. *Nat. Ecol. Evol.* **1**, 132 (2017). [doi:10.1038/s41559-017-0132](https://doi.org/10.1038/s41559-017-0132) [Medline](#)
57. S. Díaz, M. Cabido, Vive la différence: Plant functional diversity matters to ecosystem processes. *Trends Ecol. Evol.* **16**, 646–655 (2001). [doi:10.1016/S0169-5347\(01\)02283-2](https://doi.org/10.1016/S0169-5347(01)02283-2)
58. S. D'agata, D. Mouillot, L. Wantiez, A. M. Friedlander, M. Kulbicki, L. Vigliola, Marine reserves lag behind wilderness in the conservation of key functional roles. *Nat. Commun.* **7**, 12000 (2016). [doi:10.1038/ncomms12000](https://doi.org/10.1038/ncomms12000) [Medline](#)
59. D. Mouillot, N. A. J. Graham, S. Villéger, N. W. H. Mason, D. R. Bellwood, A functional approach reveals community responses to disturbances. *Trends Ecol. Evol.* **28**, 167–177 (2013). [doi:10.1016/j.tree.2012.10.004](https://doi.org/10.1016/j.tree.2012.10.004) [Medline](#)