

Capacity gaps hinder the performance of marine protected areas globally

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Abstract: Amidst a proliferation of marine protected areas (MPAs) to conserve marine resources, it is unclear whether many MPAs are being effectively and equitably managed, and how MPA management influences substantive outcomes. We developed a global database of management and fish population data (433 and 218 MPAs, respectively) to assess: 1) MPA management processes; 2) MPA impacts on fish populations, and; 3) relationships between management processes and ecological impacts. Many MPAs failed to meet thresholds for effective and equitable management processes, with widespread shortfalls in staff and financial resources. Although 71% of MPAs positively impacted fish populations, these conservation impacts were highly variable. Staff and budget capacity were the strongest predictors of conservation impacts: MPAs with adequate staff capacity had ecological impacts 2.9 times greater than MPAs with inadequate capacity. Thus, continued global expansion of MPAs without adequate investment in human and financial capacity will likely lead to sub-optimal conservation outcomes.

Keywords: Marine protected areas, Convention on Biological Diversity, impact evaluation, common pool resources, biodiversity conservation, fisheries.

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1 Awareness of human impacts upon global marine biodiversity has spurred the largest expansion
2 of marine protected areas (MPAs) in history ^{1,2}. As part of the 2011 Convention on Biological
3 Diversity (CBD) Aichi Targets, 193 countries agreed to “effectively and equitably” manage 10%
4 of coastal and marine areas within marine protected areas and “other effective area-based
5 conservation measures” by 2020 ³. A 10% conservation target for MPAs has also been included
6 within Goal 14 of the United Nations Sustainable Development Goals (SDGs) ⁴. Yet despite
7 recent advances towards these coverage targets (currently 4.1% ²), the efficacy and equity of
8 many MPAs remain uncertain ²; evidence suggests that MPAs often fail to deliver positive social
9 and ecological outcomes ⁵⁻⁷.

10 It is assumed that MPAs that are effectively regulated and actively managed through equitable
11 and inclusive decision-making approaches are more likely to meet ecological and social goals
12 than those that are merely legislated on paper (‘paper parks’) and those with exclusionary
13 decision-making ⁸⁻¹⁰. However, research linking the efficacy and equity of MPA management
14 processes to conservation outcomes lies mostly in theory and select local-scale case studies ¹¹.
15 This is largely due to a lack of a globally representative dataset on MPA management ¹² and lack
16 of counterfactuals to infer ecological outcomes in the absence of MPAs ^{13,14}.

17 We constructed a global database of management and ecological data from 433 and 218 MPAs
18 (respectively) to document and examine linkages between MPA management processes and
19 conservation outcomes. Our dataset included MPAs from every tropical and temperate ocean
20 basin, ranging in size from 0.006 to 989,836 km², and span diverse social, political and
21 biophysical contexts. First, to assess the efficacy and equity of MPA management processes, we
22 drew on empirically-supported governance and management theories ^{10,15-17} (Supplementary
23 Table 1 and Extended Data Fig. 1) to identify key management process indicators from 433

24 MPAs. We extracted data on these indicators from three widely applied survey instruments
25 (Supplementary Table 2) that provided qualitative, Likert-scaled scores on questions posed to
26 MPA stakeholders concerning MPA management activities and capacities¹⁸. From these, we
27 defined binary thresholds for effective management based on the scoring criteria and alignment
28 with social theory (Supplementary Tables 1 and 3). Second, to measure ecological impacts
29 (n=218 MPAs), we compiled MPA outcome data extracted from published studies⁵ (n=40
30 MPAs) and transect or site level observations from unpublished regional and global datasets
31 (Supplementary Table 2 and Extended Data Fig. 2; n=178 MPAs). For the unpublished
32 ecological data, we calculated logged response ratios (lnRR) of mean fish biomass per unit area
33 inside MPA sites relative to statistically matched control sites (i.e., pre-establishment and/or
34 outside MPA; Methods). Finally, we investigated the relationship between management
35 processes and ecological impacts in 62 MPAs where both management and ecological data were
36 available. We used random forest and linear mixed effects models to identify important
37 management predictors of ecological outcomes, while accounting for other factors known to
38 impact fish responses to protection (e.g., MPA age and size^{7,19,20}; Methods and Supplementary
39 material).

40 **MPA management processes**

41 MPA management processes varied widely, with many of the 433 MPAs failing to meet
42 thresholds for effective management (Fig. 1a). While the majority of MPAs were legally
43 gazetted (79%) and had appropriate regulations regarding resource use (69%), most MPAs also
44 reportedly made little to no use of scientific monitoring (biological, social or management) to
45 inform management (13%). Many also reported limited capacity, with 65% of MPAs reporting

46 that their budget was inadequate for basic management needs and 91% stating that staff capacity
47 (sufficient (on-site) staff capacity/numbers) was inadequate or below optimum.

48 Most MPAs were state managed (80%), with the remaining either co-managed or managed by
49 non-state actors (e.g., NGOs, local communities; Fig. 1a). Inclusive decision-making
50 arrangements were reported in 51% of MPAs and were more common in shared/non-state
51 managed MPAs than those managed solely by state agencies ($p < 0.001$; Extended Data Fig. 3).

52 Management processes were largely consistent across geographic contexts (Fig. 1b). In Oceania,
53 however, devolved and inclusive management was more common and relatively few MPAs were
54 legally gazetted. Where data were available for all indicators (excluding non-state management;
55 $n = 277$ MPAs), only 21% of MPAs met more than half of the nine thresholds, and only five
56 MPAs (2%) met all nine thresholds (Supplementary Table 7). Twenty-two MPAs (8%) failed to
57 meet any of the threshold levels for effective and equitable management.

58 **MPA ecological outcomes**

59 MPAs on average had positive, but variable, impacts on fish populations. We observed positive
60 responses to protection in 71% of the 218 MPAs with fish biomass data. On average, fish
61 biomass was 1.6 times higher in MPAs than in matched non-MPA areas (average logged
62 response ratios (LnRR) = $0.47 + 0.96 \text{ SD}$). Positive responses were observed across almost all
63 geographies and habitats (Fig. 2), consistent with other analyses^{5,20}. Response ratios varied
64 marginally by latitudinal zone ($F = 2.963$, $p = 0.087$; Fig. 2b) and significantly among habitats ($F =$
65 6.403 , $p < 0.001$; Fig. 2c) and continental regions ($F = 5.284$, $p < 0.001$; Fig. 2d). MPAs or MPA
66 zones where all fishing was prohibited (“no-take”) had higher response ratios than MPAs/zones
67 where fishing was permitted (“multi-use”) by almost two-fold ($t = 2.24$, $p = 0.026$; Extended Data

68 Fig. 4). Nonetheless, on average, we observed positive response ratios in both multi-use MPAs
69 and MPA zones that prohibited fishing. Responses in prohibited fishing areas were lower than in
70 some previous studies (for example, 82% increase in fish biomass in our study vs. 387% reported
71 elsewhere ⁵), likely due in part to the statistical matching approach, which reduced the
72 observable biases arising from non-random MPA placement.

73 **Linking MPA management and outcomes**

74 We then explored the relationships between management processes and ecological impacts in
75 MPAs for which we had both management and ecological data (62 MPAs in 24 countries), while
76 accounting for other significant MPA and contextual attributes (e.g., MPA age, size, ocean
77 conditions; Supplementary Table 4). In these MPAs, adequate staff capacity was the most
78 important factor in explaining fish responses to MPA protection (Fig. 3a). Budget capacity was
79 the second most important management variable and had similar performance in other analyses
80 (Supplementary Table 9); however, budget data were only available in 43 MPAs. Clearly defined
81 boundaries, MPA age and size, location (ecoregion, country), mean chlorophyll concentration,
82 and mean shore distance were also identified as important by the conditional inference forest
83 models (Fig 3a).

84 Our results demonstrate that effective biodiversity conservation is not simply a function of
85 environmental (e.g., ocean conditions) or MPA features (e.g., MPA size, age, fishing
86 regulations), but is also heavily dependent on available capacity (Fig. 3). Staff capacity was by
87 far the most important explanatory variable in our study, accounting for approximately 19% of
88 the variation in ecological outcomes ($p < 0.001$). Qualitative examination of the MPA
89 management data indicated that additional staff resources were needed to support monitoring,
90 enforcement, administration, community engagement and sustainable tourism activities (*inter*

91 *alia*). Though specific capacity needs varied among MPAs, biomass response ratios were on
92 average 2.9 times greater in MPAs reporting adequate staff capacity than those MPAs reporting
93 inadequate or no capacity (Fig. 3b). Where data were available (n=43 MPAs), we observed a
94 significant relationship between budget capacity and ecological impacts (Supplementary Table
95 9), even after we removed potential outlying data (Extended Data Fig. 5a; n=42 MPAs; $t= 2.55$;
96 $p= 0.019$). Budget capacity was also significantly correlated with staff capacity (Spearman's rho
97 0.35, $p<0.001$), and both capacity variables were positively correlated with many of the other
98 management variables (Extended Data Fig. 6). Thus, the effectiveness of many other key
99 management processes may be limited by available human and financial capacity.

100 In addition to staff capacity, clearly defined boundaries and appropriate regulations were
101 significantly correlated with ecological outcomes (Extended Data Fig. 7). However, the
102 predictive strength of these two variables was sensitive to the modelling approach. Other
103 management variables theorized to foster sustainable outcomes in common pool resources (e.g.,
104 inclusive decision making, monitoring of the resource and users¹⁵) were not significantly related
105 to ecological performance (Fig. 3a and Supplementary Table 9), a finding consistent with some
106 previous studies^{21,22}. A possible explanation is that these described processes have stronger,
107 more direct effects on resource users than on resource conditions²² or that the indicators used in
108 management assessments may imperfectly measure the governance and management processes
109 from common pool resource theory²³ (e.g., Ostrom's design principles¹⁵).

110 Like others, we found that non-management factors such as MPA age and size also shape MPA
111 ecological impacts (Fig 3a)^{7,19,20}. Although we observed a significant difference in ecological
112 impacts between prohibited fishing and multi-use zones (Extended Data Fig. 4), fishing
113 regulations were not significant in our sample of 62 MPAs while controlling for (or interacting

114 with) other factors (Fig 3a. and Supplementary Table 9). Other variables, such as proximity to
115 shore and chlorophyll concentration (a potential proxy for ocean productivity²⁴ but also for
116 reduced coastal water quality at extremely high levels²⁵), were negatively correlated with fish
117 biomass. This suggests that land-based stressors may be shaping impacts inside nearshore MPAs,
118 as noted in other work^{25,26}. Differences in variable constructs among studies may partially
119 explain observed differences in our results from previous work. For example, a recent study that
120 found “enforcement” to be a significant factor⁷ measured the enforcement construct as a
121 combination of compliance, community support, and enforcement activities, whereas our study
122 focused on management inputs into enforcement activities.

123 **Assessing MPA efficacy and equity**

124 We drew on social theory (Supplementary Table 1) to identify aspects of MPA management
125 hypothesized to be important for ecological outcomes, independent of many of the MPA and site
126 features also known to affect MPA performance (e.g. MPA age, size^{7,19}). Our theory-based
127 analytic framework (Extended Data Fig. 1 and Supplementary Table 1) provides a robust,
128 replicable approach to measuring the procedural and substantive efficacy and equity of protected
129 areas. In particular, integrated use of impact evaluation methodologies and indicators derived
130 from widely used MPA monitoring tools permits us to make novel, evidence-based inferences of
131 conservation impacts at a global scale²⁷. Despite uneven geographic distribution and limited data
132 on some indicators, this study represents one of the most comprehensive assessments of MPA
133 management and ecological outcomes to date. While the ecological data center heavily on areas
134 in the Northern Atlantic, U.S. Pacific, and Australia, the available management data are more
135 dominant in other geographies (e.g., Africa, Europe, Southeast Asia), particularly in the
136 developing world. These spatial incongruities limit the overlap between our ecological and

137 management datasets (n=62 MPAs), but collectively provide a broad view on global MPA
138 performance.

139 Given data availability, our research focused on the efficacy and equity of MPA management
140 processes and, as an indicator of substantive efficacy, the ecological impacts of MPAs on fish
141 populations. We lacked sufficient data on other taxa to assess other ecological indicators of
142 substantive efficacy. We were also unable to measure the substantive social impacts of MPAs,
143 particularly substantive equity; the spatial and temporal resolutions of relevant data were too
144 coarse or geographically-limited to assess these impacts globally. Our research highlights a need
145 for contemporaneous social, ecological, and management data in order to fill these remaining
146 knowledge gaps and explore synergies and tradeoffs among the procedural and substantive
147 outcomes of conservation. To guide conservation policy, future research should examine
148 interactions between MPAs and other management measures (e.g., fisheries management), as
149 well as site-specific MPA capacity needs.

150 **Achieving global conservation targets**

151 As we approach the CBD and SDG milestone year of 2020, the global conservation community
152 and many governments will continue to invest heavily in MPA expansion ¹. Although many
153 MPAs with low management capacity in our sample had positive ecological impacts, in general
154 the magnitude of ecological impacts was strongly linked to the available human and financial
155 capacity for MPA management. Given the widespread shortfall in staff capacity that we
156 document worldwide (Fig. 4), inadequate capacity appears to compromise the ecological
157 performance of many MPAs. Adequate capacity is likely to be even more critical in the future, as
158 increasing anthropogenic pressures on marine resources necessitate more resilient marine
159 ecosystems and corresponding management regimes. For effective and equitable management to

160 be achieved, increased investment in MPA capacity is necessary. Rapid MPA expansion without
161 increased investment has the potential to dilute already scarce resources across a larger
162 management area, weakening management and leaving many marine habitats and species at risk.
163 With such a high dependence on under-resourced MPAs to meet current and future conservation
164 and sustainable development goals^{3,4}, investment in MPA capacity development would
165 potentially result in high returns on investment for both people and nature²⁸.

166

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226 **Figure legends**

227 **Figure 1 | Percent of MPAs exceeding or falling below threshold values for indicators of effective and equitable**
228 **management processes.** Values shown for **a**, all MPAs (n=433 MPAs) and **b**, by continent. Dark blue bars (right)
229 indicate the proportion of MPAs with scores at or above the threshold value, light blue bars (left) indicate the
230 proportion below the threshold. Details on indicators, scores and threshold values in Supplementary Tables 1 and 3.

231 **Figure 2 | Fish biomass response ratios (natural log scale) for 218 MPAs.** **a**, Global variation in ln response ratio
232 (lnRR) values. Positive response ratios (blue) indicate MPAs with greater biomass inside MPA relative to matched
233 non-MPA areas. Negative values are in red. Base map sourced from ⁵⁰. **b-d**, Mean response ratios (dot) and 95%
234 confidence interval (error bars) for multi-use areas (light blue) and areas where fishing is prohibited (dark blue) in 260
235 zones in 218 MPAs shown by **b**, latitudinal zone, **c**, habitat, and **d**, continental regions. Y-axis parentheses indicate
236 the number of MPAs/zones (multi-use, fishing prohibited respectively).

237 **Figure 3 | Relationship between MPA management processes and ecological impacts.** **a**, Random forest variable
238 importance measures for management (dark blue bars) and other (non-management; light grey bars) variables as
239 they relate to ecological impacts in 62 MPAs. Importance measures exceeding the red dashed line are considered
240 non-random. **b**, Average fish biomass response ratios (dot) and 95% confidence interval (error bars) for multi-use
241 areas (light blue) and areas where fishing is prohibited (dark blue) by reported staff capacity (excluding MPAs with
242 intermediate scores (n=4)). Y-axis parentheses indicate the number of MPAs/zones (multi-use, fishing prohibited
243 respectively). Additional bivariate plots in Extended Data Fig. 5.

244 **Figure 4 | Reported level of MPA staff capacity.** MPAs reporting adequate (dark blue), inadequate or below
245 optimum (blue) and no (light blue) staff capacity in their most recent management assessments where spatial data
246 were available (n=243 MPAs; excludes MPAs with intermediate scores (n=5)). Base map sourced from ⁵⁰.

247

248 **METHODS**

249 **MPA attribute and zone information.** MPA geospatial and attribute data (i.e., location,
250 shape/boundaries, age, area, fishing regulations) were sourced from the October 2015 version of
251 the World Database on Protected Areas (WDPA)²⁹ as well as other regional and international
252 MPA datasets (see Supplement Information). Where possible, these data were supplemented
253 and/or validated using scientific publications, reports, other official government and non-
254 government sources, the ecological data providers, and local expert knowledge (Supplementary
255 Table 4). For the purpose of this study, “fishing prohibited” refers to an MPA or zone within an
256 MPA that prohibits any type of fishing activity, including subsistence and recreational fishing.

257 **MPA management data.** Data on MPA management processes were sourced from three
258 management assessment tools: Management Effectiveness Tracking Tool (METT)³⁰, the World
259 Bank MPA Score Card³¹, and the NOAA Coral Reef Conservation Program’s (CRCP) MPA
260 Management Assessment Checklist³² (Supplementary Table 2).

261 Management indicator scores were rescaled to ensure construct validity between the assessments
262 (Supplementary Table 3). To assist with the interpretation of the different scoring levels and
263 criteria, we defined binary thresholds for each indicator based on the description of the scoring
264 levels and social theory (Supplementary Tables 1 and 3). These thresholds were for descriptive
265 purposes only; we used the rescaled indicator scores (as described in Supplementary Table 3) in
266 the statistical models. For MPAs that had multiple management assessments, we used the most
267 recent assessments available for describing the status of management processes in MPAs
268 worldwide (e.g., for results in Fig. 1). For the models testing relationships with ecological
269 outcomes, we used the assessment that was closest in time to when the ecological surveys were
270 done, preferably before the ecological data were collected. If no assessment was available before

271 the ecological surveys, we chose the one closest in time after the survey. When there was more
272 than one assessment in the same year we used the median score. There were a few cases of
273 survey respondents reporting non-integer scores (e.g. 2.5) or cases when such scores arose from
274 calculating the median value for a specific year (see Extended Data Fig. 8). No rounding was
275 carried out on non-integer scores, however; MPAs with these non-integer values were excluded
276 from maps and graphics (Fig. 3b and Fig. 4) to simplify interpretation.

277 **Ecological impact data.** We derived ecological data on marine fish populations from seven
278 independent global and regional datasets, with the majority comprising species-level data from
279 underwater visual census (UVC) surveys on coral or rocky nearshore reefs (Supplementary Table
280 2), and the remainder coming from meta-analyses (Lester et al. 2009; Lester & Halpern 2008).
281 For the UVC data (15,978 survey sites), biomass represents the total biomass of all recorded fish
282 species, averaged across all transects at each site (grams per 100m²). Variations in sample
283 methods meant that the choice of recorded species varied between datasets; therefore response
284 ratios were never calculated among surveys from different datasets. Biomass values were
285 calculated by the data providers or the authors using the individual body lengths and allometric
286 length-weight data obtained either from the data provider or from FishBase (www.fishbase.org).

287 **Isolating MPA impacts.** We identify MPA impacts by comparing MPA survey sites to
288 comparable non-MPA sites (outside MPA boundaries and/or before establishment) and
289 calculating logged response ratios (LnRR). Here we use statistical matching and other
290 procedures (described below) to account for: i) selection biases in MPA placement; ii) spatial-
291 temporal dynamics of fish response to protection (e.g. spill-over, recovery time) and; iii) other
292 biological, social and physical factors that can affect fish populations ¹⁴.

293 Effective assessment of MPA impacts necessitates the isolation of response to protection (MPA
294 treatment) from other confounding factors ³³. Statistical matching allows us to develop a
295 functional counterfactual by using the same factors that determine where MPAs are placed (e.g.
296 opportunity costs for fishing) to select control sites ^{13,14}. Other factors that explain variation in
297 fish populations (e.g., habitat, depth, wave energy) can also be used as covariates in the matching
298 process. This assumes that, conditional on confounding covariates (both observed and
299 unobserved), the control and treatment sites are inter-exchangeable, that is, from the same
300 population ³⁴. Thus, with appropriate metrics or proxies of potentially confounding variables,
301 control (non-MPA) and treatment (MPA) survey sites can be appropriately matched, with the
302 majority of the remaining variation in the differences between the two groups attributable to the
303 treatment (MPA protection) effect ³⁵.

304 **Controlling for spill-over and response time-lags.** Before matching, we removed survey sites
305 that might confound the measurement of impacts. To account for (spatial) spillover effects, only
306 control survey sites greater than one kilometer away from an established MPA boundary were
307 used in the analysis (1,116 control sites removed). Despite many individual species having larger
308 home ranges ^{36,37}, a review of studies examining spillover effects of marine reserves by Halpern
309 et al ³⁸ indicates that one kilometer is a sufficient distance beyond which most population-level
310 MPA effects can no longer be detected. Any spillover effects present in sites beyond this range
311 will result in a more conservative estimate of MPA effects as it will reduce the inside-outside
312 differences.

313 To account for time lags in fish response to protection, we assigned a survey site to an MPA only
314 if the MPA was established for at least three years. Initial detectable responses to protection can
315 be quite rapid (e.g. 1.5-2 years ³⁹, 1-3 years ⁴⁰, 2-5 years ⁴¹) and three years appeared to be

316 sufficient time for MPA effects to become detectable. All sites within an MPA less than three
317 years old were not used as MPA (treatment) sites (n=579 sites). All survey sites located within
318 the boundaries of an MPA before the first (complete) year of MPA establishment were treated as
319 “before” (control) sites given that a protection response is unlikely to occur within so short a
320 period of time (n=123 sites or 3.0% of 4,125 control sites).

321 After removing the above mentioned sites and sites with ambiguous locations (n=1,882 sites
322 total), we proceeded with matching on 14,096 survey sites, comprising 9,971 treatment (MPA)
323 and 4,125 (non-MPA) control sites.

324 **Matching to control for observable bias.** Based on existing literature on MPA site-selection
325 biases and factors affecting variation in fish populations, Supplementary Table 5 describes the
326 variables compiled for each survey site and used in the matching process. We performed
327 multivariate matching using the Matching package 4.9-0³⁵ in the statistical software R v 3.2.3⁴².
328 We assessed the performance of various matching iterations using the post-matching covariate
329 match balance outputs (Supplementary Table 6) and quantile-quantile plots. Here we attempted
330 to reduce the standardized mean differences between covariates for control (non-MPA) and
331 treatment (MPA) to below 5%, which is considered appropriate for studies assessing casual
332 inference⁴³. We chose nearest neighbor multivariate matching algorithms (based on
333 Mahalanobis distances), as they performed better than propensity score algorithms for our data.
334 As there were fewer control than treatment sites, we matched with replacement, and allowed
335 multiple control sites to be matched to each treatment site. Matching with replacement prevents
336 ordering effects and allows the algorithm to choose the best available match from the entire
337 population of control sites. Allowing multiple treatment-control matches reduces the influence of
338 outliers by increasing the number of matched pairs. For our data, matching two controls to each

339 treatment site (2:1 ratio) resulted in lower standardized mean differences in treatment-control
340 covariates than 1:1 matching, or using higher ratios (e.g. 3:1,4:1). All covariates carried equal
341 weight, however covariate ‘calipers’ were used to ensure lower differences between the
342 treatment and control sites for select covariates ¹⁴ (see Supplementary Table 5). To help
343 determine appropriate calipers, we used random forest models and partial dependency plots to
344 explore the relationship between each covariate and fish biomass (using no-take sites to control
345 for fishing effects). These were useful in determining both the strength of the relationship
346 between the covariate and fish biomass, and to identify asymptotic peaks beyond which the
347 covariate has no effect (e.g. shore distance appeared to have little effect on fish biomass beyond
348 20 km). Calipers improved the quality of the matching, but reduced the overall number of
349 possible matches; 2,335 (23%) treatment (MPA) sites were dropped due to failure to find
350 appropriate controls to match the treatment sites. Some of these drops were due to failure to find
351 an appropriate control site within the same country or close in time to match with the treatment
352 site. This resulted in 15,821 matched pairs for 7,636 treatment sites in 178 MPAs. These
353 matched pairs were used to derive (natural log) response ratios for total fish biomass, which were
354 averaged to the MPA level (Extended Data Fig. 8k).

355 We used Rosenbaum’s bounds sensitivity analysis to assess the vulnerability of our MPA
356 treatment effects to unobserved biases (i.e., factors not included in our list of matching covariates
357 that could confound our estimates of MPA impact ^{34,44}). Rosenbaum’s sensitivity bounds do not
358 indicate whether or not such biases exist, but merely the potential for such a bias to influence our
359 findings. When assessing the sensitivity of our estimates of MPA impacts on fish biomass to an
360 unobserved variable, we find that if such a variable was able to change the odds of a site being
361 protected by a factor (Γ) of 1.35, it would confound our estimate of impact. While $\Gamma=1.35$

362 suggests some sensitivity in our findings to potential unobserved bias, there is no evidence to
363 suggest such a bias exists. Our extensive list of observed covariates (Supplementary Table 5)
364 were identified through expert knowledge, the scientific literature, and available primary and
365 secondary data as key factors that affect both MPA participation and outcomes. Further,
366 covariates that remained significant after matching (e.g. shore distance, chlorophyll) were
367 controlled for in subsequent models (Supplementary Table 9).

368 We supplemented the matched UVC data (n=178 MPAs) with MPA-level fish biomass ratios
369 from the Lester et al. datasets^{5,20} (n=40 MPAs), which comprise response ratios derived from
370 149 peer-reviewed publications that examine the ecological effects of areas where fishing is
371 prohibited (marine reserves or no-take areas) and areas where fishing is allowed but restricted
372 (multi-use). Where data were available for an MPA in both the Lester et al. and matched datasets
373 (n=11 MPAs), we chose the latter. No matching was required for the Lester et al. data as
374 response ratios were already formulated by the authors in their meta-analysis. The final
375 ecological dataset totaled 218 MPAs (see Extended Data Fig. 2 for data compilation steps).

376 **Management and ecological data analysis.** We used random forests with conditional inference
377 trees⁴⁵ to identify the management processes (Supplementary Table 4) that best explained the
378 variation in ecological impacts (n=62 MPAs). Random forests account for higher-order
379 interactions and nonlinear relationships between predictors, and do not require many of the strict
380 assumptions of linear parametric models that are difficult to meet⁴⁶. These qualities make
381 random forests an ideal approach for our analysis, where many interacting and non-linear
382 relationships among management processes, MPA attributes, and ecological outcomes are
383 expected¹¹. Random forests are also able to effectively estimate variable importance in “small n,
384 large p” models and models with missing data^{46,47}.

385 In this study, we used the *R* “party” package v1.0-25⁴⁸ to estimate the relative variable
386 importance of the ten management indicators using the log fish biomass response ratios as the
387 response variable and the metric for ecological impacts. In addition to the management
388 indicators, we also included other non-management variables as predictors in the model. Many
389 of these were identified in the literature as being important in explaining variability in fish
390 populations and MPA ecological outcomes (MPA age, MPA size, fishing regulations)^{7,19,20}, and
391 include many of the variables used in the matching process (mean MPA depth, shore distance,
392 market distance, human population density, chlorophyll, wave exposure, sea surface temperature,
393 ecoregion, country; Supplementary Table 5). This allowed us to assess the relative importance of
394 the management indicators as predictors, while accounting for (and allowing interactions with)
395 these potentially important non-management factors.

396 Given that we were investigating the MPA level effects of management, the MPA was
397 considered as the unit of analysis. Therefore all variables, including response ratios, were
398 averaged to the MPA level. All non-management predictors represent the MPA-level average of
399 the conditions at each fish survey site (e.g. mean depth represents the mean depth of the fish
400 survey sites in that MPA). All continuous predictors were transformed to the natural log scale to
401 reduce the effect of extreme outliers with the exception of depth which did not need to be
402 transformed. Proportion no-fishing represents the proportion of survey sites for an MPA sampled
403 from within a prohibited-fishing (no-take) zone (0: all multi-use, 1: all prohibited fishing). See
404 the Supplemental Information for more details on the procedures and variables used in the
405 random forest modelling.

406 We also ran a series of general linear mixed-effects models (Supplementary Table 9) to examine
407 the direction and strength of the relationships between each of the management indicators and

408 ecological impacts. The linear mixed effects models allowed us to examine the predictor-
409 response relationships in a hierarchical model structure, while controlling for other important
410 non-management factors. These non-management variables were those identified as important in
411 the random forest models (mean chlorophyll, mean shore distance, mean MPA age, MPA size)
412 and those found to be important in the literature (i.e., fishing regulations: “proportion no
413 fishing”). For the hierarchical structure, we included a random intercept for country to account
414 for potential non-independence in the fish response to protection between MPAs in the same
415 country (e.g. MPAs managed by the same national agency). Including country as a random
416 intercept performed similarly to other random effect structures that account for spatial hierarchy
417 (see Supplementary Table 8). We used the *R* “nlme” package v3.1-128⁴⁹ to implement the linear
418 mixed models and only included one management predictor in each model due to strong
419 correlation (Extended Data Fig. 6) and missing data amongst some of the predictor variables.
420 The results are shown in Supplementary Table 9.

421

422 **Methods references**

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Extended data legends

Extended Data Figure 1 | Key domains and illustrative indicators for assessing management efficacy and equity. Indicators with asterisks are those that were used in this study. Details on indicator descriptions, sources and citations are located in Supplementary Table 1.

Extended Data Figure 2 | Sources and major steps in the data compilation and analysis. See Supplementary Table 2 for more details on data sources. *CRCP: Coral Reef Conservation Program

Extended Data Figure 3 | Percent of MPAs by managing authority exceeding or falling below threshold values for indicators of effective and equitable management processes. Details on indicators, scores and threshold values in Supporting Tables 1 and 3. Dark blue bars (right) indicate the proportion of MPAs with scores at or above the threshold value, light blue bars (left) indicate the proportion below the threshold. Scores are from the latest assessment year where data were available from 433 MPAs.

Extended Data Figure 4 | Mean response ratios (natural log scale) of fish biomass. Mean (dot) and 95% confidence intervals (error bars) for areas where fishing is prohibited (dark blue) and multi-use MPA areas (light blue) in 254 zones in 218 MPAs.

Extended Data Figure 5 | Relationship between MPA-averaged fish biomass response ratios and key predictor variables used in the analysis of the relationship between MPA management processes and ecological impacts (n<62 MPAs). **a-j**, mean (black point) and 95% confidence intervals (error bars) of the response ratios for each management score and indicator. Details on threshold levels and score descriptions in Supplementary Table 3. **k-t**, Smoothed LOESS lines (blue line) along with the standard error region (shaded area) for relationships with continuous variables. Number of MPAs in parentheses.

Extended Data Figure 6 | Spearman rank correlations amongst management indicators, national variables and other key variables (n=433 MPAs). Variables ordered using hierarchical clustering, displaying values for significant correlations only ($p < 0.05$). Circle size and color indicate the correlative strength and direction (blue positive and red negative) respectively. Most of the management indicators for procedural efficacy were significantly correlated with each other (e.g. correlation coefficient for monitoring and management plan = 0.49).

National level variables (GDP, HDI) were poorly correlated with management indicators and were not included in this study. ENF: Adequate enforcement; BGT: Acceptable budget capacity; REG: Appropriate MPA regulations; MON: Monitoring informing management activities; MPL: Implementing existing management plan; BND: Clearly defined boundaries; LEG: Legally gazetted; STF: Adequate staff capacity/presence; IDM: Inclusive decision-making; DEV: Non-state/shared management; SIZ: MPA size (ln(km²)); AGE: MPA age (ln(years)); HDI: Human Development Index 2010; GDP: Gross Domestic Product per capita (ln(US\$ PPP)) 2013.

Extended Data Figure 7 | Spearman rank correlations amongst fish metrics, management indicators, and

other key variables for the 62 MPAs used in the management and ecological data analysis. Circle size and

color indicate the correlative strength and direction (blue positive and red negative) respectively. Variables ordered

by type (i.e. ecological, management, etc.) and not hierarchical clusters, displaying values for significant

correlations only ($p < 0.05$). BIO: (ln) fish biomass response ratio; DEN: (ln) fish density response ratio; FSZ: (ln)

fish mean size response ratio; RCH: (ln) fish species richness response ratio; DEV: Non-state/shared management;

IDM: Inclusive decision-making; LEG: Legally gazetted; BND: Clearly defined boundaries; REG: Appropriate

MPA regulations; ENF: Adequate enforcement; MON: Monitoring informing management activities; MPL:

Implementing existing management plan; STF: Adequate staff capacity/presence; BGT: Acceptable budget capacity;

NTZ: Proportion of survey sites for an MPA sampled from within a prohibited-fishing (no-take) zone; SIZ: MPA

size (ln(km²)); AGE: MPA age (ln(years)); CHO: chlorophyll-a concentration (ln (mg/m³)); SHR: Distance from

shore (ln (km)).

Extended Data Figure 8 | Frequency distribution of MPA-management, ecological and other key variables.

White bars indicate the distribution of: **a-j**, scores from the latest available management assessments ($n \leq 433$ MPAs);

k-n, MPAs where fish biomass data were available ($n \leq 218$ MPAs). **a-n**, Grey bars indicate MPAs used in the

analysis modeling the relationship between management processes and ecological impacts ($n \leq 62$ MPAs). Indicators

for **b**, inclusive decision-making and **g**, enforcement have a maximum score of 2. Non-integer values were reported

scores by few managers, or represent the median value of multiple assessments in the latest year. **k**, average MPA

level response ratios (natural log scale) for fish biomass. **l**, proportion of survey sites for an MPA sampled from

within a prohibited-fishing (no-take) zone (0: all multi-use area; 1: all no-take/prohibited fishing area). **m**, MPA age

(years between establishment and fish survey). **n**, MPA size (000 km²). MPA age and size were transformed to the log scale for the analysis.

Extended Data Figure 9 | Random forest variable importance plots. Random forest variable importance measures for management (blue bars) and other (non-management; grey bars) variables as they relate to ecological impacts in 62 MPAs. Results from models with **a**, all management indicators (as shown in Fig. 3a in the main text) and **b**, management indicators with few missing data and not highly correlated with other predictors (i.e. excluding legal status, acceptable budget, management plan, country and ecoregion). Only values greater than the red dashed line are considered to have non-random importance scores. Also shown in **c** and **d** are the predicted and observed response ratio values from the random forest models in **a** and **b** respectively, along with the linear fitted line (dashed blue line) and a smoothed LOESS line along with the standard error region (grey line and shaded area). R² values for the linear fit are also shown.