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SUPPLEMENT ARTICLE

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Characteristics of effective marine protected areas in Hawaiʻi

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Abstract

1. Ancient Hawaiians developed sophisticated natural resource management systems that included various forms of spatial management.
2. The state of Hawaiʻi established its first legislated marine protected area (MPA) in 1953, and today there exists a patchwork of spatial marine management strategies along a range of sizes, with varying levels of governance, enforcement, and effectiveness.
3. Approximately 12% of waters within the 50 m depth contour and 5% of waters within state jurisdiction (≤3 nmi) have some form of marine management. No-take areas make up <0.5% of nearshore waters, and combined with highly protected areas account for 3.4% of this habitat. Most of the existing MPAs are small, with a median area of 1.2 km^2 (confidence interval 0.2–8.1).
4. Twenty-five datasets, representing 1,031 individual surveys conducted throughout Hawaiʻi since 2000, were used to compare fish assemblage characteristics amongst a subset of MPAs using a regulation-based protection classification scheme.
5. Fully and highly protected areas had significantly greater resource fish biomass than areas with intermediate or low protection did. High human population density adjacent to MPAs had a negative influence on fish trophic structure within MPAs, whereas remote MPAs harboured higher fish biomass. Complex and heterogeneous habitats were important contributors to MPA effectiveness.
6. Long-term monitoring of select MPAs showed mixed and complex trajectories. Resource fish biomass increased after the establishment of the Hanauma Bay Marine Life Conservation District in 1967 but plateaued after ~15 years, followed by changes in assemblage structure from fish feeding and invasive species. The Pūpūkea Marine Life Conservation District, established in 1983, was expanded sevenfold in 2003 and showed dramatic increases in resource fish biomass following increased protection.
7. This information is critical to improving effectiveness of existing MPAs, helping inform ongoing efforts to implement a network of MPAs statewide, and aiding in the development of comprehensive statewide marine spatial planning.

KEYWORDS

archipelago, conservation evaluation, marine protected area, fish, fishing, reef

1 | INTRODUCTION

As a result of local and global anthropogenic stressors, coral reefs are becoming increasingly degraded worldwide (Hoegh-Guldberg et al., 2007; Hughes et al., 2017). The failure of contemporary management to halt these declines has led to a growing interest in exploring new and innovative approaches to conserving marine ecosystems. Marine protected areas (MPAs) have been shown to be a highly effective means of conserving biodiversity and managing fisheries, while also restoring and preserving overall ecosystem function (Gaines, White, Carr, & Palumbi, 2010; Lubchenco & Grorud-Colvert, 2015).

The effectiveness of MPAs can be influenced by many factors, including their size, shape, age, level of protection, and movement patterns of individual species (Babcock et al., 2010; Botsford, Micheli, & Hastings, 2003; Claudet et al., 2008; Edgar et al., 2014; Gill et al., 2017). Human impacts in surrounding areas can affect the capacity of MPAs to deliver key conservation benefits (Cinner et al., 2018). MPAs in low human-impact areas are required for sustaining ecological functions like high-order predation, but reserves in high-impact areas can provide substantial conservation gains in fish biomass. Most MPAs include a variety of zoning and management schemes, ranging from single to multiple zones and from no-take to multiple-use areas (Spalding, Fish, & Wood, 2008). Fully protected areas have been shown to have much greater conservation benefits than areas under lesser levels of protection (Horta e Costa et al., 2016; Lester et al., 2009; Sala & Giakoumi, 2017). Larger MPAs protect a greater amount and diversity of habitats, as well as critical habitats and processes that help maintain ecosystem integrity, providing protection for a wider range of species and buffering against environmental fluctuations and large-scale disturbances (Allison, Gaines, Lubchenco, & Possingham, 2003; Dayton, Sala, Tegner, & Thrush, 2000; Roberts et al., 2017). Large MPAs are also more likely to contain fully functional ecosystems and suffer less from outside effects since they have a smaller perimeter-to-area ratio (Bartholomew et al., 2008; McLeod, Salm, Green, & Almany, 2009).

Closing certain areas to harvest for periods of time has been practised for centuries by Pacific Islanders to help sustain healthy populations of marine resources (Cinner, Marnane, McClanahan, & Almany, 2006; Johannes, 1982). Hawaiians of old (pre-Western contact, before AD 1778) developed sophisticated and complex management systems for marine resource use that included various forms of spatial management (Kahaʻulelio, 2006; Malo, 1951). Today, myriad state and federal authorities manage Hawaiʻi's coastal resources, which include various forms of spatial management (Friedlander, Stamoulis, Kittinger, Drazen, & Tissot, 2014). Hawaiʻi established its first MPAs over 65 years ago, and since that time numerous MPAs have been created with varying levels of protection, ranging from complete ʻno-take' areas to areas that allow a wide variety of activities to occur within their boundaries. Designation of many of these areas was not based on comprehensive biological selection criteria or a systematic ecological assessment, but rather the existing system was built piecemeal and is reflective of various approaches to manage user conflicts, safeguard protected species, or address the wishes of local communities. A comprehensive examination of existing marine life conservation districts (MLCDs)—the most restrictive type of MPA in Hawaiʻi—previously showed that areas fully protected from fishing had higher fish biomass, larger overall fish size, and higher biodiversity than adjacent areas of similar habitat quality (Friedlander, Brown, & Monaco, 2007a, b).

Reef fish populations and their associated fisheries have declined dramatically around Hawaiʻi over the past 100 years due to a growing human population, destruction of habitat, introduction of new and unsustainable fishing techniques, and loss of traditional conservation practices (Friedlander et al., 2014; Smith, 1993). Today in Hawaiʻi, resource (food) fish biomass is low in areas near large human populations; however, remote areas with small human populations in the main Hawaiian Islands (MHI) still support high standing stock of important fisheries species (Friedlander et al., 2018; Williams et al., 2008).

Owing to the large number and variety of MPAs in Hawaiʻi, the objectives of this work were first to characterize these protected areas based on their potential for biodiversity conservation and fisheries replenishment according to existing regulations. The next objective was to examine the efficacy of these protected areas based on biophysical and governance factors, followed by an examination of the long-term trends in three well-established MPAs. This information is critical to developing sustainable fisheries management strategies, which includes improving management of existing MPAs, helping inform ongoing efforts to implement a network of MPAs statewide, and aiding the development of comprehensive marine spatial planning.

2 | METHODS

2.1 | Marine managed areas classification

Within the MHI, there are more than 90 unique marine managed areas that regulate fishing and other marine-related activities (State of Hawaiʻi Office of Planning, 2018; Figure 1, Supporting Information Table S1). These areas include MLCDs (designed primarily for conservation), fisheries management areas (designed to resolve conflicts among users), and fish replenishment areas (aquarium fish protected areas), along with various areas with other designations, such as military exclusion zones, national parks, and community-based management areas. Some of these areas were excluded from these analyses because they did not meet the criteria specific to the classification

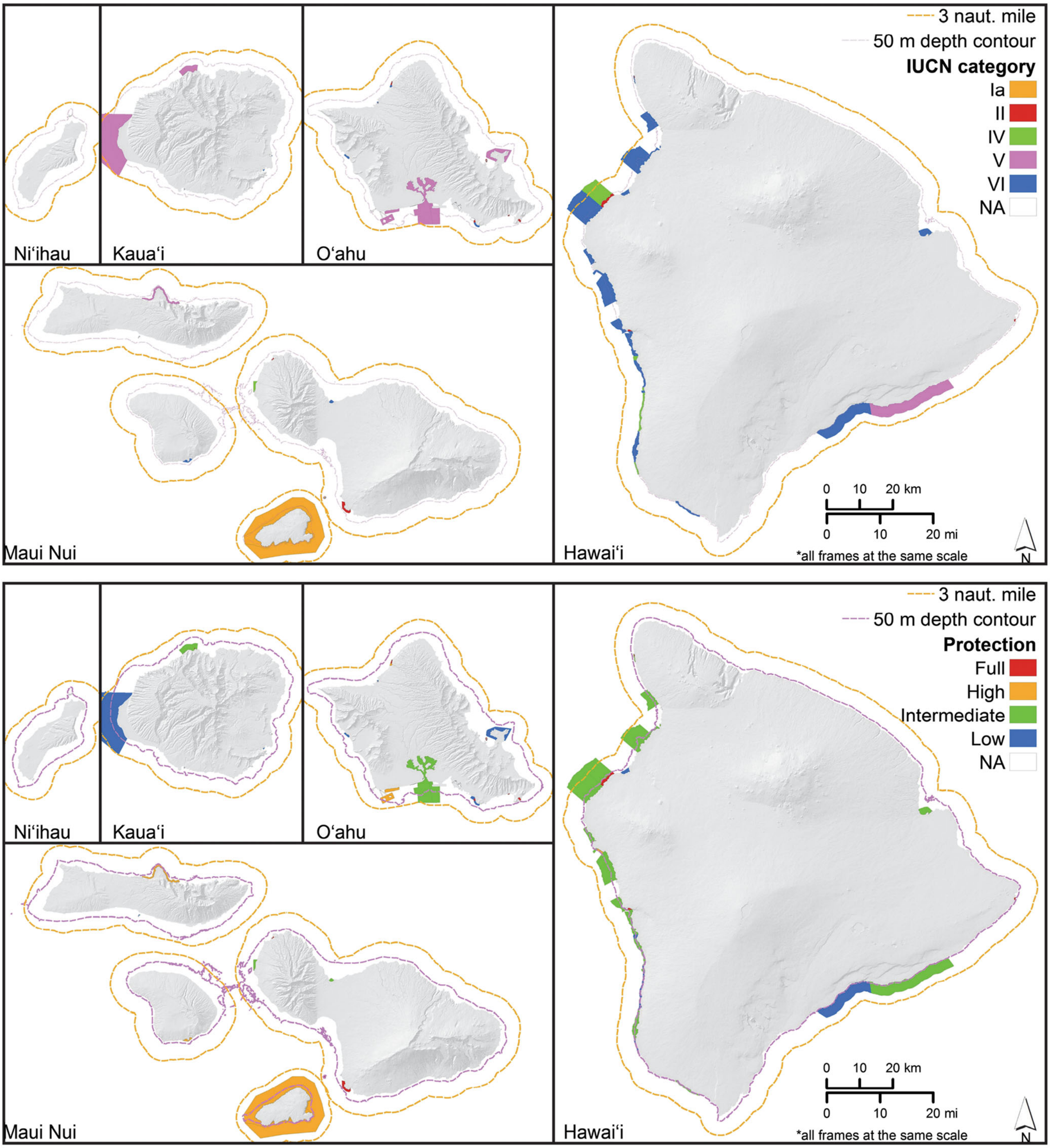

FIGURE 1 Marine protected areas in the main Hawaiian Islands mapped by International Union for the Conservation of Nature categories and protection class based on the modified regulation-based classification system of Horta e Costa et al. (2016). Dotted tan line delineates 3 nmi state waters and the dotted lavender line delineates the 50 m depth contour around each island

schemes employed (e.g. harbours, piers, anchorages, canals, specific military designations).

Because of the diversity of MPAs in Hawai'i, areas were grouped into management categories based on existing classification schemes to evaluate their effectiveness. The International Union for the Conservation of Nature (IUCN) protected area management categories are a global framework, recognized by the Convention on Biological Diversity, which distinguishes six categories of protection (i.e. scientific reserve, national park, natural monument/national landmark, nature conservation reserve, protected landscape) based on their management objectives (Dudley, 2008). MPAs were classified by the IUCN categories using guidance provided by the IUCN specific to MPAs (Table 1; IUCN World Commission on Protected Areas, 2018).

These IUCN categories were not designed to capture the variety of regulations within MPAs and did not seem appropriate to classify Hawai'i's existing MPAs for purposes of assessing protection effectiveness. An alternative classification scheme has been proposed by Horta e Costa et al. (2016), which focuses on the potential impact of different gear types and regulations on biodiversity and habitats (Table 2). The regulation-based classification system was adapted to the unique characteristics of the MPAs and nearshore fisheries found in Hawaiʻi, ultimately reducing them to four major classes of protection (full, high, intermediate, and low). Highly protected areas permit limited fishing, which is typically restricted to gear types such as pole-and-line that have low gear efficiency and minimal habitat impacts. Fully and highly protected areas were together classified as strongly protected for analysis purposes owing to the small number of fully no-take MPAs in Hawaiʻi, and previous work in Hawaiʻi (Friedlander et al., 2007a) and elsewhere (Horta e Costa et al., 2016) showing that highly protected areas can have comparable fish biomass and assemblage structure to fully protected no-take MPAs. Intermediate levels of protection allow a wider range of gear types but restrict gear types that are highly efficient and those most damaging to ecosystem function and habitats (e.g. gill nets, scuba spear). Low levels of protection were assigned to MPAs that only regulated a specific gear (e.g. gillnets) or fishery (e.g. coastal pelagic species).

2.2 | Fish sampling methods

To compare the efficacy of these MPAs, fish assemblage structure among MPAs was examined using data from underwater visual surveys. Twenty-five datasets were compiled, representing 1,031 individual fish surveys at 26 MPAs from throughout the MHI since 2000. These data were rigorously checked for errors and integrated into a common database with a standardized structure. A number of methods were used to assess fish populations across the MHI, including belt transects of various dimensions (e.g. 25 m × 5 m, 25 m × 4 m, and 25 m × 2 m), stationary point counts (15 m diameter), and 5 min timed swims. Details of each method are described in Supporting Information Table S2.

Underwater surveys of fishes in Hanauma and Kealakekua bays were first conducted in 1952 using long (230 m × 3 m) belt transects. Following the creation of the Hanauma Bay MLCD in 1967 and the Kealakekua Bay MLCD in 1969, long-term monitoring was initiated by the Hawaiʻi Division of Aquatic Resource (HDAR) using this same long belt transect method. In 1982, the Pūpūkea MLCD was established and HDAR began long-term monitoring of fishes at this site using the same methodology. Beginning in 2005, HDAR changed their sampling

TABLE 1 Characteristics of marine protected areas in Hawaiʻi based on IUCN categories (IUCN World Commission on Protected Areas, 2018)

Characteristics	Ia	II	IV	V	VI	Total
Number	3	12	4	10	41	70
Total area (km^2)	202.4	9.9	38.5	312.5	308.8	872.1
Total area 50 m (km^2)	44.6	9.9	3.5	5.8	120.8	184.6
Area ≤50 m (%)	2.0	0.5	0.2	0.3	5.5	8.4
Area ≤3 nmi (%)	1.2	0.1	0.2	1.9	1.9	5.2
Area, mean (SD) (km^2)	67.5 (116.5)	0.8 (1.2)	9.6 (17.9)	31.2 (0.6)	7.5 (42.1)	12.5 (31.3)
Area, median (95% CI) (km^2)	0.4 (0.1–202.0)	0.3 (0.1–0.7)	1.0 (0.1–27.7)	11.9 (7.4–41.6)	1.3 (0.3–6.9)	1.6 (0.3–8.2)
Age (years), mean (SD)	23.7 (18.6)	39.2 (13.8)	10.7 (7.6)	32.2 (20.2)	28.5 (9.8)	29.7 (12.9)

TABLE 2 Characteristics of marine protected areas in Hawaiʻi based on based on the modified regulation-based classification system of Horta e Costa et al. (2016)

Characteristics	Full	High	Intermediate	Low	Total
Number	12	6	40	14	72
Total area (km^2)	10.2	221.5	412.5	228.0	872.2
Total area 50 m (km^2)	10.1	62.7	126.6	70.7	270.1
Area ≤50 m (%)	0.5	2.9	5.8	3.2	12.3
Area ≤3 nmi (%)	<0.1	1.3	2.5	1.4	5.2
Area mean (SD) (km^2)	0.8 (1.2)	36.9 (80.9)	10.3 (17.6)	16.3 (36.6)	12.1 (30.9)
Area, median (95% CI) (km^2)	0.3 (0.2–0.7)	4.7 (1.1–56.3)	3.3 (0.4–9.1)	0.6 (0.1–14.4)	1.2 (0.2–8.1)
Age (years), mean (SD)	36.1 (16.9)	32.8 (13.9)	26.3 (11.3)	27.7 (7.9)	28.9 (12.6)

methods within these MLCDs to be consistent with the West Hawai‘i Aquarium Project (25 m × 2 m), while also adding a 5 min timed swim for resource species (Supporting Information Table S1).

2.3 | Methods calibration

Underwater visual fish survey methods each have unique biases (Colvocoresses & Acosta, 2007; Edgar, Barrett, & Morton, 2004; McClanahan et al., 2007). Differences in the performance of these survey methods requires that data gathered by multiple methods should be standardized before being combined for analysis (Maunder & Punt, 2004). To account for overall differences in survey methods, conversion factors were calculated to standardize methods using general linear models and Monte Carlo simulations to calculate methods calibration factors (Nadon, 2014). Calibrations were calculated by species, where possible, using the following decision rules: (1) $\geq$10 paired observations were available within an island; (2) if the proportion of zeros was high (>15%), a delta model was run where occurrences were modelled separately from non-occurrences; (3) the fit was checked for normally distributed residuals, and if this check failed the model was rerun and checked with log-transformed data. If a species did not pass this series of rules then a calibration factor was not calculated, and a calibration factor for each combination of family and trophic level was calculated and applied instead. If a calibration factor could not be calculated at the combined family–trophic level, then a global calibration was used that considered all species pooled for each method. For all subsequent analyses, density estimates were based on calibrated densities of raw data (Friedlander et al., 2018, table S3).

2.4 | Biomass estimates

The biomass of individual fishes was estimated using the allometric length–weight conversion: $W = a \times \text{TL}^b$, where parameters a and b are species-specific constants, TL (cm) is total length, and W (g) is weight. These constants were obtained from a comprehensive assessment of Hawai‘i length–weight fitting parameters (Froese & Pauly, 2011). The cross-product of individual weights and numerical densities was used to estimate biomass by species. Fishes were categorized into four trophic groups (piscivores, invertivores, planktivores, and herbivores) after DeMartini, Friedlander, Sandin, and Sala (2008) and Sandin et al. (2008). Resource species were defined either as those species having $\geq$450 kg of average annual commercial or recreational harvest from 2000 to 2010 or as recognized species that are important to the local subsistence or cultural sectors (Friedlander et al., 2018).

2.5 | Statistical analyses

Total fish biomass and resource fish biomass among major MPA classes (full and high combined, intermediate, and low) were analysed using generalized linear mixed-effects models (GLMMs, Bolker et al., 2009) with the ‘glmer’ function in the ‘lme4’ package in R version 3.4.1 (Bates, Mächler, Bolker, & Walker, 2015). MPA class was treated as a fixed effect. Dataset was included as a random effect to account for differences in sampling and methods across datasets included in the analyses (Friedlander et al., 2018). Model fits were assessed by visual inspection of residuals. The models for total fish biomass and resource fish biomass were fitted using a gamma error structure with a log link function (Crawley, 2012). GLMMs were rerun with an additional random effect of year and compared with models without year using likelihood ratio tests. To examine the influence of depth on MPA class, the GLMM was rerun using MPA class, depth, and their interaction as fixed effects. Hypothesis tests for fixed effects were based on likelihood-ratio tests using the ‘Anova’ function in the ‘car’ package in R (Fox & Weisberg, 2011). Unplanned post hoc multiple comparisons to discriminate between pairwise treatments of fish assemblage characteristics among classes were tested using a Tukey's honestly significant difference (HSD) test in the ‘glht’ function in the ‘multcomp’ v1.4–7 package in R (Hothorn, Bretz, & Westfall, 2008).

To describe the pattern of fish trophic structure within MPAs and their relationship to MPA characteristics, direct gradient analysis (redundancy analysis, RDA) was performed using the ordination program CANOCO version 5.0 (Ter Braak & Šmilauer, 2012). Linear models were appropriate, as a preliminary detrended correspondence analysis showed short gradient lengths along the ordination axes (<2 SD). The response variables were centred and log transformed fish trophic biomass data by MPA. Explanatory variables consisted of level of protection, MPA age, MPA size, benthic habitat characteristics (PC1, PC2), and human population density associated with each MPA. All explanatory variables were centred and standardized prior to analysis. Pearson's product-moment correlation analysis was used to test for dependence between explanatory variables, which resulted in no pairwise comparisons greater than ρ = 0.54. We therefore included all explanatory variables in the RDA. To rank MPA characteristics in their relevance to fish assemblage structure, interactive forward selection was used; the statistical significance of each variable was judged by a Monte Carlo unrestricted permutation test with 499 permutations (ter Braak & Verdonschot, 1995).

Benthic habitat characteristics were generated by extracting detailed structure data (e.g. rock/boulder, aggregated coral reef, pavement, sand, patch reef) from benthic habitat maps derived from visual interpretation of multispectral IKONOS and Quickbird satellite imagery (Battista, Costa, & Anderson, 2007). Habitat boundaries were delineated around signatures (e.g. areas with specific colour and texture patterns) in the orthorectified imagery mosaic corresponding to habitat types in the classification scheme. Thematic map accuracy was >90.0% for all detailed habitat structures. Although these vector-based benthic habitat maps are limited to a two-dimensional planar representation, they are the best available for the study area and adequately represent the reef surface area except in areas with great topographic change (e.g. steep slopes and reef walls), which were not common in the MPAs that were assessed.

A principal components analysis was performed using the area of coverage for detailed structure categories among MPAs. PC1 and PC2 from the benthic principal components analysis were used as variables to describe the benthic community among MPAs in the RDA. Human population density was used as a proxy for isolation and human impacts associated with MPAs. Total human population based on 2010 census data (www.census.hawaii.gov) within each *moku* (traditional Hawaiian district) was divided by the shoreline length of that *moku* to provide an index of human population pressure (Friedlander et al., 2018).

Long-term trends in resource fish biomass were examined for the Hanauma Bay and Pūpūkea MLCDs on Oʻahu, and zones A and B of the Kealakekua Bay MLCD on Hawaiʻi Island. All fishing is prohibited within Kealakekua zone A. Within zone B, it is permitted to fish for, take, or possess any finfish using hook-and-line and thrownet. These long-term datasets have been collected by the Hawaiʻi Division of Aquatic Resources starting in 1952 in Hanauma Bay and Kealakekua Bay zones A and B, and in Pūpūkea starting in 1983. GLMMs were fitted for Hanauma Bay, Pūpūkea, and Kealakekua zone A. Significance of the fixed effect (survey year) was tested using the F-statistic. Survey method was included as a fixed effect to account for the influence on density estimation due to detection differences. Each transect site was treated as a random effect to account for repeated surveys. A simple linear model was used to test the annual trend for Kealakekua zone B since there was only one transect site and one method. Segmented regression (Muggeo, 2008) was also used to explore non-linear trends in resource fish biomass across years in each MLCD using the R package ʻsegmentedʼ and approximate F-test using Kenward–Rogers approximation for degrees of freedom in the R package ʻlmerTestʼ.

Fish assemblage size spectra (Graham, Dulvy, Jennings, & Polunin, 2005; Rochet & Trenkel, 2003) were described for each long-term MLCD using least-squares regression to relate $\log_{10}$-transformed numerical densities to body length in 3 cm size bins. Size spectra were performed for each year, with the resulting slopes plotted against years and fitted to least-squares regressions. F-statistics were used to examine trends in slopes for each MLCD using the ʻcarʼ package in R. Years with less than five size bins were eliminated from these analyses (Kealakekua Bay zone A: 1952; Kealakekua Bay zone B: 1970, 1994, 1998, 1999, 2003).

To examine patterns in fish assemblage structure over time in each long-term MLCD, constrained analysis of principal coordinates with a Bray–Curtis distance matrix was performed using the ʻveganʼ package in R on fish biomass by family using year as the constraint. Biomass of each family for each transect was standardized using Wisconsin double standardization. The misclassification error or residual error was used to obtain a non-arbitrary decision concerning the appropriate number of dimensions to include in the constrained analysis of principal coordinates (Anderson & Willis, 2003).

3 | RESULTS

3.1 | Characteristics of marine managed areas

Most of the existing marine management areas are small, with a median area of 1.2 km^2 (confidence interval 0.2–8.1). The Kahoʻolawe Island Reserve is the largest MPA in the MHI at 202.1 km^2, which accounts for 23% of total MPA coverage within state waters (3 nmi from shore) and 16% of waters within the 50 m depth stratum. The mean age of MPAs is 28.9 years (±12.6 years SD), with the oldest being the Moku o Loʻe Marine Laboratory Reserve, established in 1953.

The total area considered under marine management was 872 km^2, which covered 5% of state waters within 3 nmi from shore. Of this, 270.1 km^2 was within the 50 m depth contour and encompassed 12.6% of this nearshore area using the regulation-based classification scheme. One-third of the MPAs in the database could not be classified using the IUCN criteria. Of the remaining areas, IUCN Categories Ia and II (the most restrictive designations) together covered 2.5% of waters within the 50 m depth stratum.

Owing to the difficulties in assigning IUCN categories to many of the MPAs in Hawaiʻi, the regulated-based scheme was used to assess MPA contributions to marine protection. Under the regulation-based classification scheme, fully protected areas made up <0.5% of nearshore waters (within 50 m depth stratum) and combined with highly protected areas accounted for 3.4% of the nearshore area. However, most of this coverage was within the Kahoʻolawe Island Reserve, which incorporates 42.2 km^2 of highly protected nearshore waters. If Kahoʻolawe was excluded from this calculation, then <1% of nearshore waters were highly protected.

The MPAs of Hawaiʻi Island comprised 35.9% of all nearshore areas under marine management, with 33% of the island protected in some manner (Table 3). An additional 24% of nearshore protected area occurs around Oʻahu, although <2% of this is classified as strongly protected. The island of Kauaʻi contributed an additional 18.2% to total protected area within 50 m depth; however, 81% of this coverage is located within the Barking Sands Pacific Missile Testing Range Danger Zone (39.9 km^2), which provides low protection. The entire island of Kahoʻolawe has a high level of protection under the Kahoʻolawe Island Reserve. For all other islands, strong levels of protection of nearshore waters ($\leq$50 m depth) range from ~2.5% of the nearshore area around Molokaʻi to virtually none around Kauaʻi.

3.2 | MPA efficacy

Resource fish biomass (g m^{-2}) was highest in Kalaupapa National Historical Park ($\overline{X}$ = 164.8 ± 117.8 SD) and the Kahoʻolawe Island Reserve ($\overline{X}$ = 125.7 ± 102.3 SD), and lowest at the Waikiki Beach Restricted Area ($\overline{X}$ = 2.8 ± 2.0 SD, Figure 2). There were significant differences in total fish biomass (Gamma GLMM: $x^2_{2,\ 1,021} = 112.1$, $P < 0.01$) and resource fish biomass (Gamma GLMM: $x^2_{2,\ 1,021} = 126.7$, $P < 0.01$) among the three levels of regulation-based MPA protection. In both cases, fish biomass was significantly higher ($P < 0.01$) in highly protected areas than in areas of intermediate or low levels of protection, and significantly higher in intermediate areas than in areas of low protection (high > intermediate > low). No effect of survey year was found when modelled independently against resource fish biomass (Gamma GLMM, $P = 0.65$), suggesting little change in biomass within MPAs over the

TABLE 3 Area (km^2) of levels of protection by island, based on the modified regulation-based classification system of Horta e Costa et al. (2016). Nearshore = area ≤ 50 m depth

Island	Statewide nearshore area protected (%)	Area (km^2) of nearshore ≤ 50 m depth	Full	High	Intermediate	Low	Total	Nearshore protected by island (%)
Hawai'i	35.9	295.6	4.6	1.0	73.7	17.6	96.9	32.8
Kaho'olawe	16.4	44.2	—	44.2	—	—	44.2	100.0
Kaua'i	18.2	332.8	0.1	—	9.2	39.9	49.2	14.8
Lana'i	0.4	131.0	—	1.2	—	0.0	1.2	0.9
Maui	2.4	366.1	3.8	—	2.8	—	6.5	1.8
Moloka'i	2.9	306.7	—	7.6	—	0.1	7.8	2.5
O'ahu	23.0	531.0	1.7	8.6	40.9	13.0	64.3	12.1
Total	100.0		10.1	62.7	126.6	70.7	270.1	

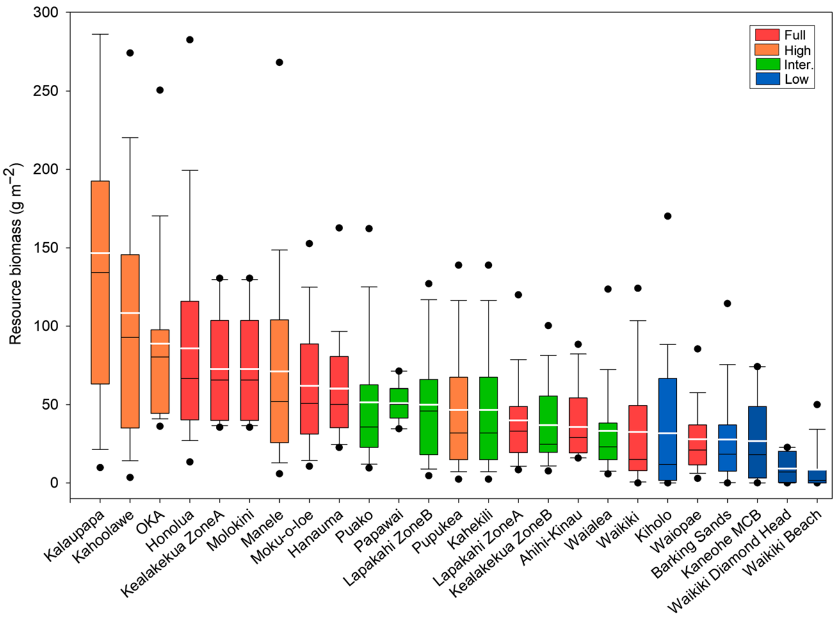

FIGURE 2 Resource fish biomass (g m^{-2}) for marine protected areas (MPAs) in Hawai'i, ordered from highest to lowest. Box plots showing median (black line), mean (white line), upper and lower quartiles, and 5th and 95th percentiles. MPAs are colour coded according to protection class. Inter. = intermediate

study period. There was a significant effect of depth on resource fish biomass (Gamma GLMM, $P < 0.001$), but the interaction of depth and MPA class was not significant (Gamma GLMM, $P = 0.09$), indicating that depth was not a significant determinant of differences in MPA class efficacy.

MPAs were well separated in ordination space according to regulated-based levels of protection (Figure 3). The first two axes of the RDA biplot explained 62.0% of the variance among fish trophic groups and 98.1% of the trophic groups and MPA variables relationship (Table 4). MPAs with high levels of protection explained 22.6% of the variation in fish trophic structure, followed by PC2, which explained 22.0% of the variation. PC2 was a proxy for habitat type, with the major loading being scattered coral/rock (0.73), rubble (0.73), aggregated reef (0.65), and rock/boulder (−0.62). MPAs with low protection from fishing explained an additional 12.8% of the variation, followed by population density by *moku* (district), which accounted for an additional 6%.

3.3 | Temporal comparisons

There were significant increases in resource fish biomass in Hanauma Bay ($F_{1,\ 576} = 8.2$, $P < 0.01$) and Pūpūkea ($F_{1,\ 354} = 8.8$, $P < 0.01$) MLCDs over the study period (Figure 4). Kealakekua zones A ($F_{1,\ 215} = 2.7$, $P = 0.10$) and B ($F_{1,\ 32} = 2.9$, $P = 0.10$) showed no apparent trends over time. Segmented linear regression showed a significant increase (slope: 8.6; $F_{1,\ 33} = 14.7$, $P < 0.01$) in resource fish biomass in Hanauma Bay after initial protection, with a break point in 1982, after which time there has been a slight decrease in resource fish biomass (slope: −1.04; $F_{1,\ 536} = 30.5$, $P < 0.01$). However, there was a significant effect of survey method, which changed in 2005 ($P < 0.01$). The initial slope of resource fish biomass in the Pūpūkea MLCD was not significant (slope: 0.17; $F_{1,\ 316} = 0.8$, $P = 0.37$) but increased dramatically after 2015 (slope: 25.8; $F_{1,\ 79} = 8.8$, $P < 0.01$). Initial trends in resource fish biomass in Kealakekua Bay zone A were relatively flat (slope: 0.01; $F_{1,\ 35} = 0.48$, $P = 0.49$), with a slight but non-significant

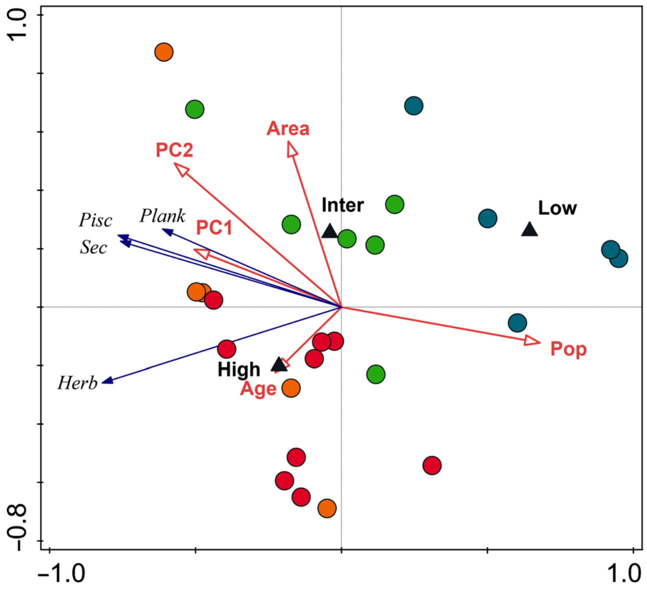

FIGURE 3 Biplot of results of redundancy analysis on fish biomass of trophic groups with marine protected area (MPA) variables (level of protection, size, age, human population by *moku*, PC1, and PC2). Data were centred and log-transformed fish biomass for trophic groups by MPA. MPA characteristics were centred and standardized prior to analysis. Pisc: piscivores; Invert: invertivores; Plank: planktivores; Herb: herbivores. MPAs are colour coded according to protection class: red circles, fully protected; brown circles, highly protected; green circles, intermediate protection (Inter); and blue circles, low protection

TABLE 4 (a) Results of redundancy analysis (RDA) on log-transformed fish biomass data for trophic groups with MPA variables (level of protection, size, age, human population by *moku*, PC1, and PC2). (b) Conditional effects of Monte-Carlo permutation results on the RDA

(a) Axes	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalues	0.58	0.04	0.01	0.01
Explained variation (cumulative)	58.35	62.04	63.01	63.22
Pseudo-canonical correlation	0.88	0.51	0.43	0.21
Explained fitted variation (cumulative)	92.29	98.12	99.66	100.00
(b) Variables	Explained (%)	Contribution (%)	Pseudo- F	P
High protection	22.6	32.7	6.7	0.006
PC2	22.0	31.9	8.7	0.006
Low protection	12.8	18.6	6.3	0.004
Human population density	5.8	8.4	3.2	0.034

decline (slope: −1.0; $F_{1,\,177} = 3.70$, $P = 0.06$) after a break point in 1996. However, there was a modestly significant effect of survey method ($P = 0.03$). Segmented regression could not be performed on Kealekekua zone B since the years around the break point were not surveyed adequately for the regression to be fit.

Size spectra analysis was used to examine changes in size structure of fish assemblages in each MLCD over time. Slopes of the size spectra showed significant increases in Hanauma Bay and Kealakekua Bay zone B ($P < 0.01$ for both) since establishment (Figure 5). However, in Kealakekua zone A and Pūpūkea the slopes of their size spectra were not significantly different from zero since establishment.

All MLCDs showed significant family compositional changes over time (Hanauma: $F_{1,\,581} = 28.68$, $P = 0.001$; Pūpūkea: $F_{1,\,439} = 6.78$, $P = 0.001$; Kealakekua Bay zone A: $F_{1,\,218} = 9.92$, $P = 0.001$; Kealakekua Bay zone B: $F_{1,\,33} = 5.51$, $P = 0.001$) (Figure 6). Constrained variables (time) explained 25% of the variation at Kealakekua zone A, 62% at Kealakekua zone B, 8% at Pupukea, and 11% at Hanauma. The misclassification error or residual error accounted for 50.6% of the residual variation at Kealakekua zone A, 28.5% at Kealakekua zone B, 65.8% at Pupukea, and 61.7% at Hanauma.

In most cases, within-decade concordance was high. In the early years (1960–1970s) in Hanauma Bay, goatfishes (Mullidae) and parrotfishes (Scaridae) were most prevalent. In the late 1970s, chubs (Kyphosidae) showed a strong influence until the late 1990s, which likely resulted from an increase in fish feeding during that time. Once fish feeding was banned in 1999, the family composition reverted to something resembling that in the 1960s, but with the addition of two invasive species from the families Serranidae (*Cephalophus argus*) and Lutjanidae (*Lutjanus kasmira*).

In the earlier years of the Pūpūkea MLCD, wrasses (Labridae) and surgeonfishes (Acanthuridae) were most abundant. Following increased protection in 2003, resource species such as jacks (Carangidae) and parrotfishes became more important in the assemblage. In both zones in Kealakekua Bay, wrasses were most abundant in the earlier years, followed by an increase in the importance of parrotfishes, as well as grouper and snappers, which were represented by the two invasive species (*C. argus* and *L. kasmira*). In Kealakekua Bay zone B, surgeonfishes also became more important in the latter years.

4 | DISCUSSION

The large number of MPAs in Hawai'i gives the impression of a substantial network of actively managed areas, but in reality the majority of these areas are small, and nearly all allow some form of fishing within their boundaries. The existing MPAs were built ad hoc, reflecting the need to manage user conflicts, conserve biodiversity, or address local interests. The diversity of MPAs in Hawai'i provides a unique opportunity to examine the effectiveness of existing areas while also providing guidance for future protected areas.

4.1 | Level of protection

Fully and highly protected MPAs in Hawai'i harboured greater fish biomass than areas with intermediate or low levels of protection. Currently, only 3.4% of Hawai'i's nearshore areas within 50 m depth (an

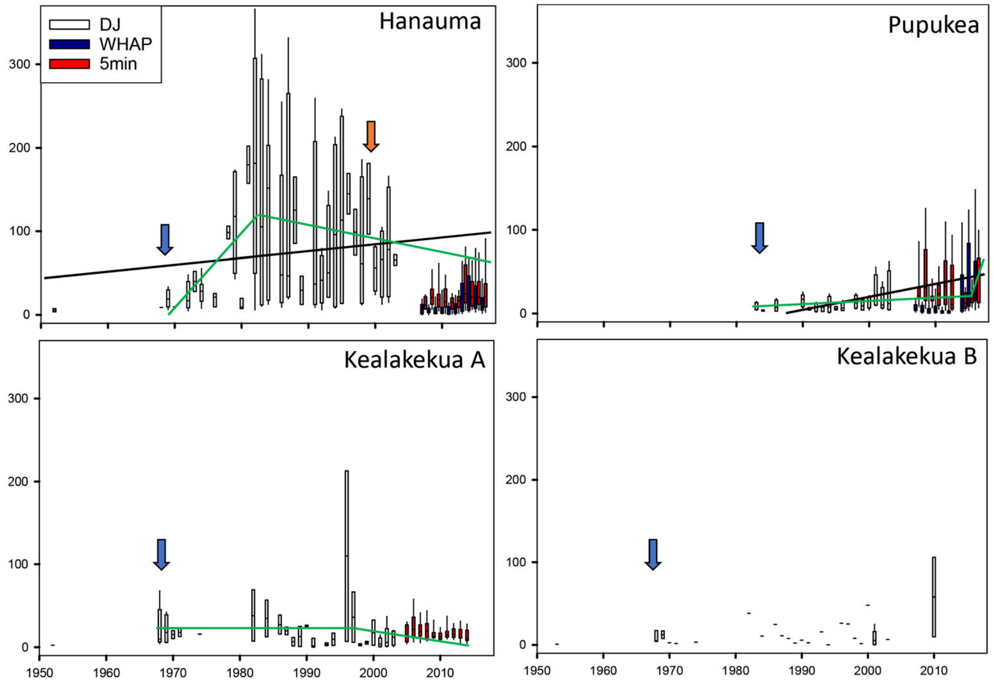

FIGURE 4 Resource fish biomass density (g m^{-2}) by survey year for three marine life conservation districts (MLCDs). Blue arrows indicate the year each protected area was established. The brown arrow shows the year fish feeding was banned at Hanauma Bay. Surveys methods include Dingell–Johnson 230 m × 3 m belt transects (DJ), 5 min timed resource fish swims (5min), and four 25 m belt transects at permanent sites (West Hawaii Aquarium Project, WHAP). Black lines show biomass trends standardized for the DJ method for MLCDs with significant increases. Green lines show fitted segmented regressions

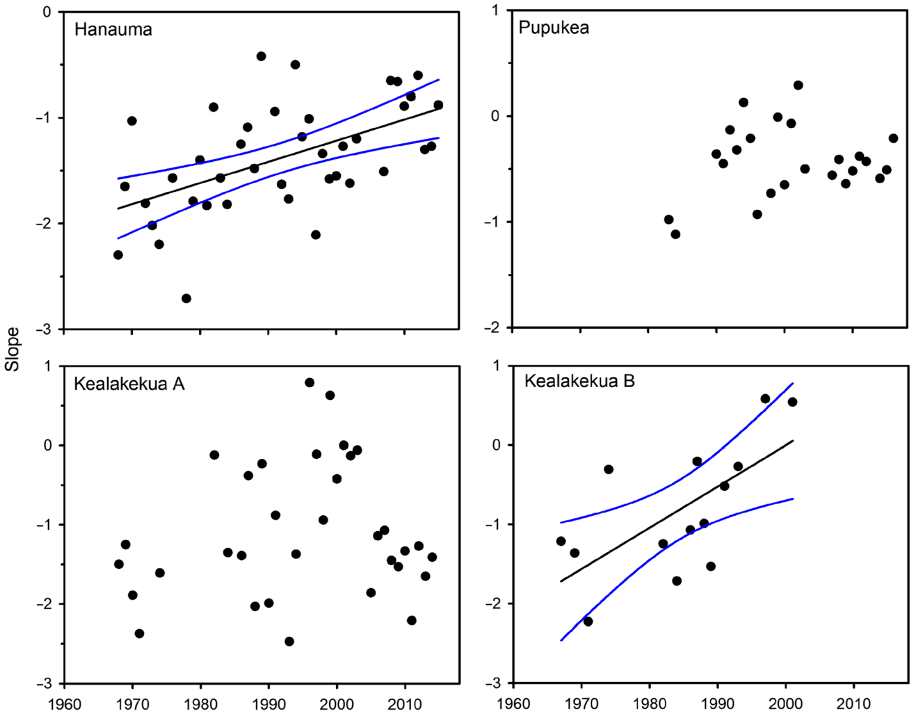

FIGURE 5 Mean slopes of size spectra for among years for three marine life conservation districts (MLCDs). Black lines show the fitted regression line for MLCDs with significant trends. Blue bands show 95% confidence intervals for the fitted regression lines

approximation of inshore habitats, which are the primary targets for fishing of reef and reef-associated species) are in strongly protected (full and high combined) areas, and only 0.5% are within no-take MPAs ('full' protection). A large majority (72%) of the state's strongly protected waters are in the Kahoʻolawe Island Reserve, and therefore the remaining extent of strongly protected areas is extremely limited.

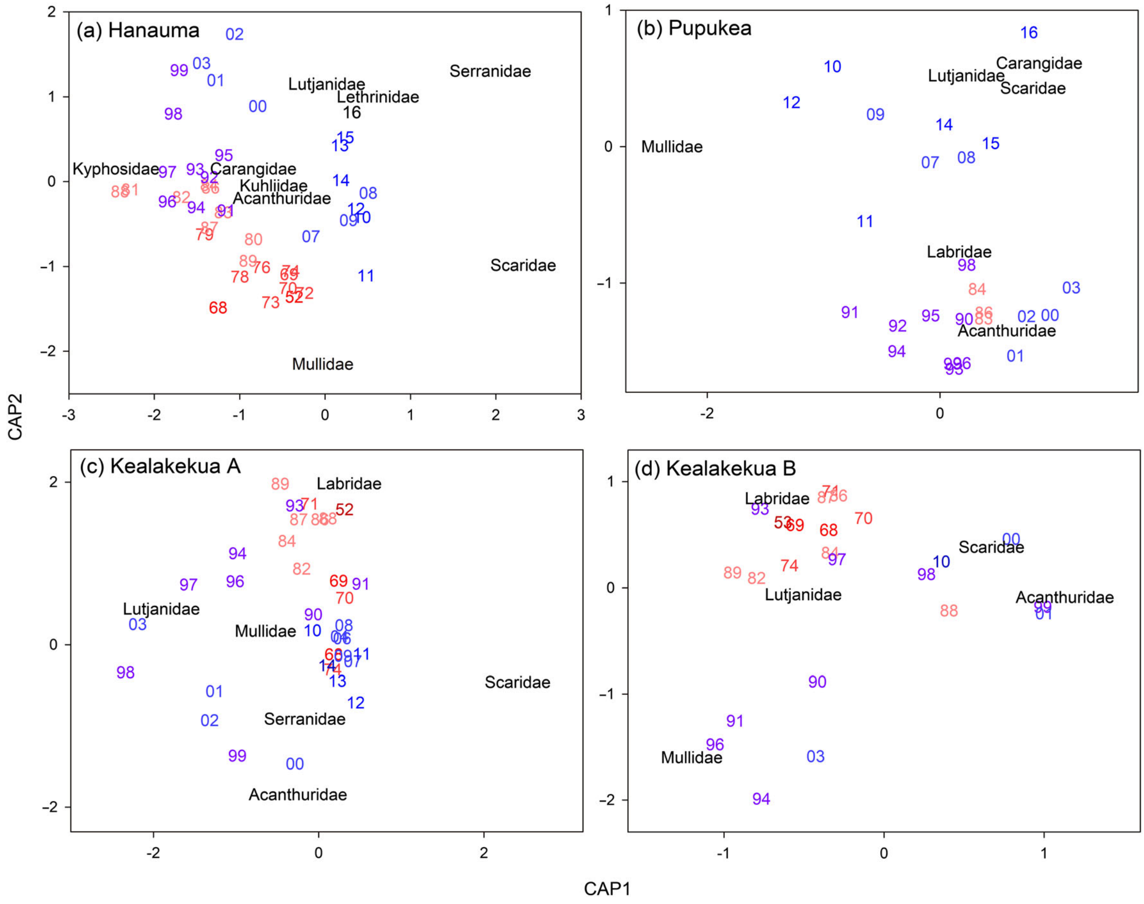

FIGURE 6 Constrained analysis of principal coordinates (CAP) of fish family biomass among years for each marine life conservation district. Only families with CAP1 or CAP2 scores with absolute values ≥0.5 are shown. Years are colour coded from red to blue/purple as older years to more recent years

Approximately 95% of state-managed waters (within 3 nmi of shore) are not spatially managed for fishing or specially restricted in any way.

The assessment of progress towards ocean conservation targets varies considerably (Fitzsimons, 2011; Horta e Costa et al., 2016). Major disagreements as to the actual area protected globally results in a false sense of accomplishment, especially when greater protection is needed (Sala & Giakoumi, 2017). The World Database on Protected Areas (United Nations Environment World Conservation Monitoring Centre & IUCN, 2018) states that ~7% of the ocean is protected, whereas analysis by the Atlas of Marine Protection (Marine Conservation Institute, 2018) suggests that only 3.6% of the world's oceans are in MPAs and only half of that is strongly protected in no-take marine reserves. The term 'MPA' is now being used so loosely that it no longer connotes meaningful protection, emphasizing the current overestimation of marine protection and highlighting the need for more transparent methods of accessing accurate marine protection levels (Malta Declaration, 2017).

4.2 | Biophysical factors

The median size of MPAs in Hawai'i (1.1 km^2) is minuscule compared with the geographic extent of the species they are designed to protect. This is even more pronounced in full and highly protected areas, where the median size is only 0.6 km^2 (confidence interval 0.3–3.4). The influence of MPA size in explaining fish trophic structure within MPAs in Hawai'i was low (~2%) and may partially be attributed to the small size of the vast majority of these areas.

Larger MPAs are more effective because they encompass biologically connected and diverse ecosystems, allowing a greater fraction of fish populations to remain protected than in smaller MPAs (Jennings, 2000; Sale et al., 2005). To maintain healthy populations of targeted fisheries species, marine reserves should be larger than the home ranges of the species of interest (Green et al., 2014). A meta-analysis of European MPAs found that for every onefold increase in no-take MPA size, there was a 35% increase in the density of commercial fishes within the reserve (Claudet et al., 2008). In Tasmania, the largest MPAs had higher fish species richness, higher density of large fish, and larger-sized exploitable fishes than the smaller reserves did (Edgar & Barrett, 1999).

Increasing MPA size often incurs significant socio-economic costs that impede the implementation of these areas (Devillers et al., 2014). In some cases, smaller MPAs have been shown to be effective (Aburto-Oropeza et al., 2011; Russ & Alcala, 2003) and are therefore useful components in a larger MPA network. Although the current small size and extent of MPAs in Hawai'i are not meeting their stated conservation and fisheries management objectives (Friedlander et al., 2007a), they can be designed as effective components of a larger network, which is an effort that is of importance considering the challenges in creating new and larger MPAs. This may be particularly relevant in highly urbanized areas, where area protection is constrained due to conflicts among multiple users (Curley, Kingsford, & Gillanders, 2003).

Habitat was also an important contributor in explaining MPA effectiveness. MPAs with a heterogeneous mix of habitat types with high complexity (e.g. aggregated reef, scattered coral, boulder, and rubble) harboured higher fish biomass and more diverse trophic assemblages than MPAs with low habitat diversity and low complexity did. This is consistent with previous work in Hawai'i showing MPAs with high habitat complexity, high coral cover, and low macroalgae cover had higher values for most fish assemblage characteristics (Friedlander et al., 2007b).

4.3 | Temporal trends

Long-term monitoring of select MLCDs has shown mixed and complex trajectories. In addition, changes in monitoring methods have complicated the interpretation of trends within these protected areas. In Hanauma Bay, resource fish biomass increased early after the 1967 establishment of the MPA but plateaued after ~15 years, with changes in assemblage structure resulting from fish feeding and invasive species. The Pūpūkea MLCD was established in 1983 and expanded in area by sevenfold in 2003 while also expanding restrictions to prohibit most fishing. Dramatic increases in resource fish biomass and the abundance of target resource fishes (e.g. jacks and parrotfishes) have been noted since this transition. Although no significant changes have been observed in resource fish biomass within either zone of the Kealakekua Bay MLCD, increases in the size structure of fishes within zone B have occurred over time, suggestive of potential indirect effects of ecological processes such as predation (i.e. protected fish growing bigger, resulting in heavier predation on smaller size classes).

Decadal-scale observations of no-take MPAs have shown that direct effects on target species typically occur within 5 years of establishment, but these effects can be vary greatly depending on the life histories of individual species (Babcock et al., 2010; Barrett, Edgar, Buxton, & Haddon, 2007; Russ & Alcala, 2004). Indirect effects can take a decade or more to develop (Babcock et al., 2010), and many non-fishery species do not respond to protection at all (Barrett et al., 2007). Species- and size-related competitive and predatory control have been noted in older MPAs, resulting in an ecological succession of dominance among different families of fishes (McClanahan, Graham, Calnan, & MacNeil, 2007). MPA size can also play an interacting role in the rates of recovery inside MPAs. In eastern Australia, many of the targeted taxa examined were more abundant in large no-take MPAs within a few years of their establishment compared with small no-take MPAs (Malcolm, Jordan, Creese, & Knott, 2016). Collectively, these studies show that MPA effects can be slow, complex, and species specific.

4.4 | Climate change

Hawai'i experienced its first statewide mass-bleaching event in 2015, and projections for coral bleaching associated with climate change in the state indicate that this disturbance will happen regularly by 2042 (Hughes et al., 2017; van Hooidonk, Maynard, Manzello, & Planes, 2014). The state of Hawai'i developed a coral bleaching recovery plan following the mass-bleaching event to support coral recovery. The plan calls for the establishment of a network of permanent no-take MPAs and a network of herbivore fishery management areas to enhance the ability of Hawai'i's reefs to resist and recover from increasingly frequent climate disturbances (Chung et al., 2019).

Across the tropical Western Pacific, modelled climate variables (e.g. sea surface temperature, dissolved oxygen, pH, and net primary productivity) projected that the maximum potential catch of coral reef fisheries would decrease >50% in many areas, with some species becoming locally extinct (Asch, Cheung, & Reygondeau, 2018). In the absence of more detailed understanding of how the biological sensitivities of reef species across their life stages interact with climate exposure directly, as well as higher order interactions within the ecosystem, a precautionary approach is warranted. In this sense, MPAs can serve to maintain intact coral reef ecosystem structure and function to better withstand mounting climate pressures. Areas where habitats and species are known to have withstood environmental changes (or extremes) in the past are potential climate change refugia and should be protected within future MPAs, because they are likely to be important for maintaining biodiversity and ecosystem function in the face of climate change (Green et al., 2014; McLeod et al., 2012).

Well-managed MPAs may help marine ecosystems and people adapt to the impacts of climate change (Roberts et al., 2017). Pacific Islanders and their knowledge–practice–belief systems have a long history of resilience to environmental variability and unpredictability, including periodic and severe disturbances (e.g. droughts, floods, storms, and tsunamis; McMillen et al., 2014). Integrating traditional ecological knowledge and customary practices into contemporary marine management has shown promise in many locations, including Hawai'i (Cinner, 2005; Friedlander, Shackeroff, & Kittinger, 2013; Kittinger, Cinner, Aswani, & White, 2014). Hybridization of customary beliefs and institutions with modern management concepts such as MPAs and ecosystem-based management can help address broader concerns, such as climate change and coastal degradation (Aswani & Ruddle, 2013).

4.5 | Marine spatial planning

The seas around Hawai'i provide commercial, recreational, and subsistence opportunities, and are vital to the state's approximately US$800 million a year marine tourism industry. Over 80% of the 9 million tourists annually visiting Hawai'i participate in some type of marine activity (Cesar & van Beukering, 2004). The economic value of the Hanauma Bay MPA alone was estimated at US$650 million in 2002 (van Beukering & Cesar, 2004). Hawai'i's nearshore resources also are important for the continuance of Native Hawaiian subsistence fishing and associated socio-cultural practices.

The most recent no-take reserve in Hawaiʻi was established in 2003, and this area was minuscule in size (0.3 km^2). There is strong opposition to the creation of additional MPAs in Hawaiʻi by the large and vocal fishing community. Fishing is an integral part of the local culture, and fishers often view MPAs as having a direct negative impact on their activities and livelihoods. This issue can partially be addressed through better public relations and educational efforts. There is much evidence to show that protected areas in fact increase spillover of fishable biomass into nearby waters (Halpern, Lester, & Kellner, 2009; Vandeperre et al., 2011), supporting local fisheries. In addition, managers need to engage the fishing community in an equitable stakeholder participatory approach. For example, incorporating fishers' local knowledge into designing MPAs has proven to be effective in gaining public support for some of these areas (Aswani et al., 2012). Given the large number of existing MPAs in Hawaiʻi, one strategy might be to increase protection within existing MPAs, which may not be as contentious as the creation of entirely new protected areas.

There is increased interest among communities and coastal stakeholders in integrating aspects of Native Hawaiian knowledge systems and customary practices into contemporary management in Hawaiʻi (Ayers & Kittinger, 2014). Hybrid systems that incorporate elements of customary and contemporary management can overcome some of the limitations to implementation of successful MPAs. Despite the numerous obstacles to formal governmental authorization, numerous communities in Hawaiʻi are strengthening local influence and accountability for their marine resources, oftentimes independent of government support (Friedlander et al., 2013).

Whether an MPA is successful in meeting the goals of supporting biodiversity and ecosystem function depends on the level of compliance and buy-in from the people that use the resource. Consistently, studies have found that human population density negatively impacts reef fish biomass throughout Hawaiʻi, though the magnitude and uncertainty of these impacts vary locally (Gorospe et al., 2018; Williams et al., 2008). Although there is considerable variation in the predicted natural capacity of different areas throughout Hawaiʻi to support reef fish biomass, human population density remains a dominant driver of decreased reef fish biomass (Friedlander et al., 2018; Gorospe et al., 2018). This clear message highlights that marine management and public-awareness efforts have a tremendous potential impact on the health of Hawaiiʻs coral reef ecosystem.

More than half of the 11,000 MPAs in the MPAtlas.org database are <10 km^2 in size, with the median size being 3.8 km^2 (Marine Conservation Institute, 2018; United Nations Environment World Conservation Monitoring Centre & IUCN, 2018). If marine reserves and other MPAs are to provide significant conservation benefits to species, they must be scaled up. Networks of MPAs provide an option for increasing the benefits often provided by single MPAs (Grorud-Colvert et al., 2014), while simultaneously achieving conservation and fishery goals (Gaines et al., 2010).

At the IUCN World Conservation Congress held in Hawaiʻi in 2016, Governor Ige of Hawaiʻi announced an initiative to effectively manage 30% of Hawaiʻi's nearshore waters by 2030. This evaluation of the effectiveness of existing MPAs in the state is an important step in implementing this 30 × 30 initiative, as it helps to identify which MPAs have been successful in Hawaiʻi in the past and to define criteria and principles necessary for effective marine conservation in Hawaiʻi in the future.

This work has broad regional and global importance owing to the large number of MPAs examined and the scale of this assessment. Typically, only a few MPAs are examined in a single study except for meta-analyses, where the results can be confounded by differences in biogeography and other factors. Our results are consistent with other study findings and can be useful in guiding similar efforts elsewhere around the world. This information is critical to improving effectiveness of existing MPAs, helping inform ongoing efforts to implement future MPAs, and aiding in the development of comprehensive marine spatial planning worldwide.

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REFERENCES

Aburto-Oropeza, O., Erisman, B., Galland, G. R., Mascareñas-Osorio, I., Sala, E., & Ezcurra, E. (2011). Large recovery of fish biomass in a no-take marine reserve. *PLoS ONE*, *6*, e23601. https://doi.org/10.1371/journal.pone.0023601

Allison, G. W., Gaines, S. D., Lubchenco, J., & Possingham, H. P. (2003). Ensuring persistence of marine reserves: Catastrophes require adopting an insurance factor. *Ecological Applications*, *13*, 8–24. https://doi.org/10.1890/1051-0761(2003)013[0008:EPOMRC]2.0.CO;2

Anderson, M. J., & Willis, T. J. (2003). Canonical analysis of principal coordinates: A useful method of constrained ordination for ecology. *Ecology*, *84*, 511–525. https://doi.org/10.1890/0012-9658(2003)084[0511:CAOPCA]2.0.CO;2

Asch, R. G., Cheung, W. W. L., & Reygondeau, G. (2018). Future marine ecosystem drivers, biodiversity, and fisheries maximum catch potential in Pacific Island countries and territories under climate change. *Marine Policy*, *88*, 285–294. https://doi.org/10.1016/j.marpol.2017.08.015

Aswani, S., Christie, P., Muthiga, N. A., Mahon, R., Primavera, J. H., Cramer, L. A., … Hacker, S. (2012). The way forward with ecosystem-based management in tropical contexts: Reconciling with existing management systems. *Marine Policy*, *36*, 1–10. https://doi.org/10.1016/j.marpol.2011.02.014

Aswani, S., & Ruddle, K. (2013). Design of realistic hybrid marine resource management programs in Oceania. *Pacific Science*, *67*, 461–476. https://doi.org/10.2984/67.3.11

Ayers, A. L., & Kittinger, J. N. (2014). Emergence of co-management governance for Hawai'i coral reef fisheries. *Global Environmental Change*, *28*, 251–262. https://doi.org/10.1016/j.gloenvcha.2014.07.006

Babcock, R. C., Shears, N. T., Alcala, A. C., Barrett, N. S., Edgar, G. J., Lafferty, K. D., … Russ, G. R. (2010). Decadal trends in marine reserves reveal differential rates of change in direct and indirect effects. *Proceedings of the National Academy of Sciences of the United States of America*, *107*, 18256–18261. https://doi.org/10.1073/pnas.0908012107

Barrett, N. S., Edgar, G. J., Buxton, C. D., & Haddon, M. (2007). Changes in fish assemblages following 10 years of protection in Tasmanian marine protected areas. *Journal of Experimental Marine Biology and Ecology*, *345*, 141–157. https://doi.org/10.1016/j.jembe.2007.02.007

Bartholomew, A., Bohnsack, J. A., Smith, S. G., Ault, J. S., Harper, D. E., & McClellan, D. B. (2008). Influence of marine reserve size and boundary length on the initial response of exploited reef fishes in the Florida Keys National Marine Sanctuary, USA. *Landscape Ecology*, *23*, 55–65. https://doi.org/10.1007/s10980-007-9136-0

Bates, D., Mächler, M., Bolker, B., & Walker, S. (2015). Fitting linear mixed-effects models using lme4. *Journal of Statistical Software*, *67*, 1–48. https://doi.org/10.18637/jss.v067.i01

Battista, T. A., Costa, B. M., & Anderson, S. M. (2007). *Shallow-water benthic habitats of the main eight Hawaiian Islands*. NOAA Technical Memorandum NOS NCCOS 61, Biogeography Branch. Silver Spring, MD: NOAA.

Bolker, B. M., Brooks, M. E., Clark, C. J., Geange, S. W., Poulsen, J. R., Stevens, M. H. H., & White, J. S. S. (2009). Generalized linear mixed models: A practical guide for ecology and evolution. *Trends in Ecology & Evolution*, *24*, 127–135. https://doi.org/10.1016/j.tree.2008.10.008

Botsford, L. W., Micheli, F., & Hastings, A. (2003). Principles for the design of marine reserves. *Ecological Applications*, *13*, 25–31. https://doi.org/10.1890/1051-0761(2003)013[0025:pftdom]2.0.co;2

Cesar, H. S., & van Beukering, P. (2004). Economic valuation of the coral reefs of Hawai'i. *Pacific Science*, *58*, 231–242. https://doi.org/10.1353/psc.2004.0014

Chung, A. E., Wedding, L. M., Green, A. L., Friedlander, A. M., Goldberg, G., Meadows, A., & Hixon, M. A. (2019). Building coral reef resilience through spatial herbivore management. *Frontiers in Marine Science*, *6*, 98. https://doi.org/10.3389/fmars.2019.00098

Cinner, J. (2005). Socioeconomic factors influencing customary marine tenure in the Indo-Pacific. *Ecology and Society*, *10*, 36. Retrieved from http://www.jstor.org/stable/26267728. https://doi.org/10.5751/ES-01364-100136

Cinner, J., Marnane, M. J., McClanahan, T. R., & Almany, G. R. (2006). Periodic closures as adaptive coral reef management in the Indo-Pacific. *Ecology and Society*, *11*, 31. Retrieved from http://www.jstor.org/stable/26267789. https://doi.org/10.5751/ES-01618-110131

Cinner, J. E., Maire, E., Huchery, C., MacNeil, M. A., Graham, N. A., Mora, C., … Mouillot, D. (2018). Gravity of human impacts mediates coral reef conservation gains. *Proceedings of the National Academy of Sciences of the United States of America*, *115*, E6116–E6125. https://doi.org/10.1073/pnas.1708001115

Claudet, J., Osenberg, C. W., Benedetti-Cecchi, L., Domenici, P., García-Charton, J. A., Pérez-Ruzafa, Á., … Culioli, J. M. (2008). Marine reserves: Size and age do matter. *Ecology Letters*, *11*, 481–489. https://doi.org/10.1111/j.1461-0248.2008.01166.x

Colvocoresses, J., & Acosta, A. (2007). A large-scale field comparison of strip transect and stationary point count methods for conducting length-based underwater visual surveys of reef fish populations. *Fisheries Research*, *85*, 130–141. https://doi.org/10.1016/j.fishres.2007.01.012

Crawley, M. J. (2012). *The R book*. Chichester, UK: John Wiley & Sons.

Curley, B. G., Kingsford, M. J., & Gillanders, B. M. (2003). Spatial and habitat-related patterns of temperate reef fish assemblages: Implications for the design of marine protected areas. *Marine and Freshwater Research*, *53*, 1197–1210. https://doi.org/10.1071/mf01199

Dayton, P. K., Sala, E., Tegner, M. J., & Thrush, S. (2000). Marine reserves: Parks, baselines, and fishery enhancement. *Bulletin of Marine Science*, *66*, 617–634.

DeMartini, E. E., Friedlander, A. M., Sandin, S. A., & Sala, E. (2008). Differences in fish-assemblage structure between fished and unfished atolls in the northern Line Islands, central Pacific. *Marine Ecology Progress Series*, *365*, 199–215. https://doi.org/10.3354/meps07501

Devillers, R., Pressey, R. L., Grech, A., Kittinger, J. N., Edgar, G. J., Ward, T., & Watson, R. (2014). Reinventing residual reserves in the sea: Are we favouring ease of establishment over need for protection? *Aquatic Conservation: Marine and Freshwater Ecosystems*, *25*, 480–504. https://doi.org/10.1002/aqc.2445

Dudley, N. (Ed.) (2008). *Guidelines for applying protected area management categories*. Gland, Switzerland: IUCN. https://doi.org/10.2305/IUCN.CH.2008.PAPS.2.en

Edgar, G. J., & Barrett, N. S. (1999). Effects of the declaration of marine reserves on Tasmanian reef fishes, invertebrates and plants. *Journal of Experimental Marine Biology and Ecology*, *242*, 107–144. https://doi.org/10.1016/S0022-0981(99)00098-2

Edgar, G. J., Barrett, N. S., & Morton, A. J. (2004). Biases associated with the use of underwater visual census techniques to quantify the density and size-structure of fish populations. *Journal of Experimental Marine Biology and Ecology*, *308*, 269–290. https://doi.org/10.1016/j.jembe.2004.03.004

Edgar, G. J., Stuart-Smith, R. D., Willis, T. J., Kininmonth, S., Baker, S. C., Banks, S., … Thomson, R. J. (2014). Global conservation outcomes depend on marine protected areas with five key features. *Nature*, *506*, 216–220. https://doi.org/10.1038/nature13022

Fitzsimons, J. A. (2011). Mislabeling marine protected areas and why it matters—A case study of Australia. *Conservation Letters*, *4*, 340–345. https://doi.org/10.1111/j.1755-263X.2011.00186.x

Fox, J., & Weisberg, S. (2011). *An R companion to applied regression* (2nd ed.). Thousand Oaks, CA: Sage.

Friedlander, A. M., Brown, E., & Monaco, M. E. (2007b). Defining reef fish habitat utilization patterns in Hawaii: Comparisons between marine protected areas and areas open to fishing. *Marine Ecology Progress Series*, *351*, 221–233. https://doi.org/10.3354/meps07112

Friedlander, A. M., Brown, E. K., & Monaco, M. E. (2007a). Coupling ecology and GIS to evaluate efficacy of marine protected areas in Hawaii. *Ecological Applications*, *17*, 715–730. https://doi.org/10.1890/06-0536

Friedlander, A. M., Donovan, M. K., Stamoulis, K. A., Williams, I. D., Brown, E. K., Conklin, E. J., … Walsh, W. J. (2018). Human-induced gradients of reef fish declines in the Hawaiian Archipelago viewed through the lens of traditional management boundaries. *Aquatic Conservation: Marine* *and Freshwater Ecosystems*, *28*, 146–157. https://doi.org/10.1002/aqc.2832

Friedlander, A. M., Shackeroff, J. M., & Kittinger, J. N. (2013). Customary marine resource knowledge and use in contemporary Hawai'i. *Pacific Science*, *67*, 441–460. https://doi.org/10.2984/67.3.10

Friedlander, A. M., Stamoulis, K. A., Kittinger, J. N., Drazen, J., & Tissot, B. N. (2014). Understanding the scale of marine protection in Hawai'i: From community-based management to the remote northwestern Hawaiian Islands Marine National Monument. *Advances in Marine Biology*, *69*, 153–203. https://doi.org/10.1016/B978-0-12-800214-8.00005-0

Froese, R., & Pauly, D. (2011). FishBase. www.fishbase.org [accessed 9 Sep 2017].

Gaines, S. D., White, C., Carr, M. H., & Palumbi, S. R. (2010). Designing marine reserve networks for both conservation and fisheries management. *Proceedings of the National Academy of Sciences of the United States of America*, *107*, 18286–18293. https://doi.org/10.1073/pnas.0906473107

Gill, D. A., Mascia, M. B., Ahmadia, G. N., Glew, L., Lester, S. E., Barnes, M., … Fox, H. E. (2017). Capacity shortfalls hinder the performance of marine protected areas globally. *Nature*, *543*, 665–669. https://doi.org/10.1038/nature21708

Gorospe, K. D., Donahue, M. J., Heenan, A., Gove, J. M., Williams, I. D., & Brainard, R. E. (2018). Local biomass baselines and the recovery potential for Hawaiian coral reef fish communities. *Frontiers in Marine Science*, *5*, 162. https://doi.org/10.3389/fmars.2018.00162

Graham, N. A. J., Dulvy, N. K., Jennings, S., & Polunin, N. V. C. (2005). Size-spectra as indicators of the effects of fishing on coral reef fish assemblages. *Coral Reefs*, *24*, 118–124. https://doi.org/10.1007/s00338-004-0466-y

Green, A. L., Fernandes, L., Almany, G., Abesamis, R., McLeod, E., Aliño, P. M., … Pressey, R. L. (2014). Designing marine reserves for fisheries management, biodiversity conservation, and climate change adaptation. *Coastal Management*, *42*, 143–159. https://doi.org/10.1080/08920753.2014.877763

Grorud-Colvert, K., Claudet, J., Tissot, B. N., Caselle, J. E., Carr, M. H., Day, J. C., … Walsh, W. J. (2014). Marine protected area networks: Assessing whether the whole is greater than the sum of its parts. *PLoS ONE*, *9*, e102298. https://doi.org/10.1371/journal.pone.0102298

Halpern, B. S., Lester, S. E., & Kellner, J. B. (2009). Spillover from marine reserves and the replenishment of fished stocks. *Environmental Conservation*, *36*, 268–276. https://doi.org/10.1017/S0376892910000032

Hoegh-Guldberg, O., Mumby, P. J., Hooten, A. J., Steneck, R. S., Greenfield, P., Gomez, E., … Knowlton, N. (2007). Coral reefs under rapid climate change and ocean acidification. *Science*, *318*, 1737–1742. https://doi.org/10.1126/science.1152509

Horta e Costa, B., Claudet, J., Franco, G., Erzini, K., Caro, A., & Gonçalves, E. J. (2016). A regulation-based classification system for marine protected areas (MPAs). *Marine Policy*, *72*, 192–198. https://doi.org/10.1016/j.marpol.2016.06.021

Hothorn, T., Bretz, F., & Westfall, P. (2008). Simultaneous inference in general parametric models. *Biometrical Journal*, *50*, 346–363. https://doi.org/10.1002/bimj.200810425

Hughes, T. P., Barnes, M. L., Bellwood, D. R., Cinner, J. E., Cumming, G. S., Jackson, J. B., … Palumbi, S. R. (2017). Coral reefs in the Anthropocene. *Nature*, *546*, 82–90. https://doi.org/10.1038/nature22901

International Union for Conservation of Nature World Commission on Protected Areas. (2018). Applying IUCN's global conservation standards to marine protected areas (MPA). Delivering effective conservation action through MPAs, to secure ocean health & sustainable development. Version 1.0. Gland, Switzerland: IUCN

Jennings, S. (2000). Patterns and prediction of population recovery in marine reserves. *Reviews in Fish Biology and Fisheries*, *10*, 209–231. https://doi.org/10.1023/A:1016619102955

Johannes, R. E. (1982). Traditional conservation methods and protected marine areas in Oceania. *Ambio*, *11*, 258–261.

Kaha'ulelio, D. (2006). *Ka 'Oihana Lawai'a: Hawaiian fishing traditions*. Honolulu, HI: Bishop Museum Press and Awaiaulu Press.

Kittinger, J. N., Cinner, J. E., Aswani, S., & White, A. T. (2014). Back to the future: Integrating customary practices and institutions into comanagement of small-scale fisheries. In J. N. Kittinger, L. McClenachan, K. B. Gedan, & L. K. Blight (Eds.), *Marine historical ecology in conservation: Applying the past to manage for the future* (pp. 135–160). Berkeley, CA: University of California Press.

Lester, S. E., Halpern, B. S., Grorud-Colvert, K., Lubchenco, J., Ruttenberg, B. I., Gaines, S. D., … Warner, R. R. (2009). Biological effects within no-take marine reserves: A global synthesis. *Marine Ecology Progress Series*, *384*, 33–46. https://doi.org/10.3354/meps08029

Lubchenco, J., & Grorud-Colvert, K. (2015). Making waves: The science and politics of ocean protection. *Science*, *350*, 382–383. https://doi.org/10.1126/science.aad5443

Malcolm, H. A., Jordan, A., Creese, R. G., & Knott, N. A. (2016). Size and age are important factors for marine sanctuaries: Evidence from a decade of systematic sampling in a subtropical marine park. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *26*, 1090–1106. https://doi.org/10.1002/aqc.2600

Malo, D. (1951). *Hawaiian Antiquities: Mo'olelo Hawai'i*. Honolulu, HI: Bishop Museum Press.

Malta Declaration: Assessing real progress towards effective ocean protection. Presented at our Oceans October 6, 2017. Retrieved from https://www.nationalgeographic.org/projects/pristine-seas/malta-declaration/

Marine Conservation Institute. (2018). MPAtlas [On-line]. Seattle, WA: Marine Conservation Institute. Available from www.mpatlas.org [Accessed (downloaded 11/04/2018].

Maunder, M. N., & Punt, A. E. (2004). Standardizing catch and effort data: A review of recent approaches. *Fisheries Research*, *70*, 141–159. https://doi.org/10.1016/j.fishres.2004.08.002

McClanahan, T. R., Graham, N. A. J., Calnan, J. M., & MacNeil, M. A. (2007). Toward pristine biomass: Reef fish recovery in coral reef marine protected areas in Kenya. *Ecological Applications*, *17*, 1055–1067. https://doi.org/10.1890/06-1450

McClanahan, T. R., Graham, N. A. J., Maina, J., Chabanet, P., Bruggemann, J. H., & Polunin, N. V. C. (2007). The influence of instantaneous variation on estimates of coral reef fish populations and communities. *Marine Ecology Progress Series*, *340*, 221–234. https://doi.org/10.3354/meps340221

McLeod, E., Green, A., Game, E., Anthony, K., Cinner, J., Heron, S., & Klepas, J. (2012). Integrating climate and ocean change vulnerability into conservation planning. *Coastal Management*, *40*, 651–672. https://doi.org/10.1080/08920753.2012.728123

McLeod, E., Salm, R., Green, A., & Almany, J. (2009). Designing marine protected area networks to address the impacts of climate change. *Frontiers in Ecology and the Environment*, *7*, 362–370. https://doi.org/10.1890/070211

McMillen, H. L., Ticktin, T., Friedlander, A., Jupiter, S. D., Thaman, R., Campbell, J., … Apis-Overhoff, L. (2014). Small islands, valuable insights: Systems of customary resource use and resilience to climate change in the Pacific. *Ecology and Society*, *19*, 44. https://doi.org/10.5751/ES-06937-190444

Muggeo, V. M. R. (2008). segmented: An R package to fit regression models with broken-line relationships. *R News*, *8/1*, 20–25.

Nadon, M. O. (2014). Improving stock assessment capabilities for the coral reef fishes of Hawaii and the Pacific region (PhD thesis). University of Miami, Coral Gables, FL.

Roberts, C. M., O'Leary, B. C., McCauley, D. J., Cury, P. M., Duarte, C. M., Lubchenco, J., … Worm, B. (2017). Marine reserves can mitigate and promote adaptation to climate change. *Proceedings of the National Academy of Sciences of the United States of America*, *114*, 6167–6175. https://doi.org/10.1073/pnas.1701262114

Rochet, M. J., & Trenkel, V. M. (2003). Which community indicators can measure the impact of fishing? A review and proposals. *Canadian Journal of Fisheries and Aquatic Sciences*, *60*, 86–99. https://doi.org/10.1139/f02-164

Russ, G. R., & Alcala, A. C. (2003). Marine reserves: Rates and patterns of recovery and decline of predatory fish, 1983–2000. *Ecological Applications*, *13*, 1553–1565. https://doi.org/10.1890/01-5341

Russ, G. R., & Alcala, A. C. (2004). Marine reserves: Long-term protection is required for full recovery of predatory fish populations. *Oecologia*, *138*, 622–627. https://doi.org/10.1007/s00442-003-1456-4

Sala, E., & Giakoumi, S. (2017). No-take marine reserves are the most effective protected areas in the ocean. *ICES Journal of Marine Science*, *75*, 1166–1168. https://doi.org/10.1093/icesjms/fsx059

Sale, P., Cowen, R., Danilowicz, B., Jones, G. P., Kritzer, J., Lindeman, K., … Steneck, R. S. (2005). Critical science gaps impede use of no-take fishery reserves. *Trends in Ecology & Evolution*, *20*, 74–80. https://doi.org/10.1016/j.tree.2004.11.007

Sandin, S. A., Smith, J. E., DeMartini, E. E., Dinsdale, E. A., Donner, S. D., Friedlander, A. M., … Pantos, O. (2008). Baselines and degradation of coral reefs in the northern Line Islands. *PLoS ONE*, *3*, e1548. https://doi.org/10.1371/journal.pone.0001548

Smith, M. K. (1993). An ecological perspective on inshore fisheries in the main Hawaiian Islands. *Marine Fisheries Review*, *55*, 34–49.

Spalding, M. D., Fish, L., & Wood, L. J. (2008). Toward representative protection of the world's coasts and oceans—Progress, gaps, and opportunities. *Conservation Letters*, *1*, 217–226. https://doi.org/10.1111/j.1755-263X.2008.00030.x

State of Hawaiʻi, Office of Planning. (2018). Retrieved from http://planning.hawaii.gov/[16 May 2018]

ter Braak, C. J. F., & Šmilauer, P. (2012). *Canoco reference manual and user's guide: Software for ordination. Version 5*. Ithaca, NY: Microcomputer Power.

ter Braak, C. J. F., & Verdonschot, P. F. M. (1995). Canonical correspondence analysis and related multivariate methods in aquatic ecology. *Aquatic Sciences*, *57*, 255–289. https://doi.org/10.1007/bf00877430

United Nations Environment World Conservation Monitoring Centre & IUCN. (2018). Marine protected planet [On-line], [March, 2018]. Cambridge, UK: UNEP-WCMC and IUCN. Retrieved from www.protectedplanet.net

van Beukering, P., & Cesar, H. (2004). *Economic analysis of marine managed areas in the main Hawaiian Islands*. Honolulu, HI: University of Hawaii.

van Hooidonk, R., Maynard, J. A., Manzello, D., & Planes, S. (2014). Opposite latitudinal gradients in projected ocean acidification and bleaching impacts on coral reefs. *Global Change Biology*, *20*, 103–112. https://doi.org/10.1111/gcb.12394

Vandeperre, F., Higgins, R. M., Sánchez-Meca, J., Maynou, F., Goni, R., Martín-Sosa, P., … D'Anna, G. (2011). Effects of no-take area size and age of marine protected areas on fisheries yields: A meta-analytical approach. *Fish and Fisheries*, *12*, 412–426. https://doi.org/10.1111/j.1467-2979.2010.00401.x

Williams, I. D., Walsh, W. J., Schroeder, R. E., Friedlander, A. M., Richards, B. L., & Stamoulis, K. A. (2008). Assessing the importance of fishing impacts on Hawaiian coral reef fish assemblages along regional-scale human population gradients. *Environmental Conservation*, *35*, 261–272. https://doi.org/10.1017/S0376892908004876

SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

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