LETTER

Conservation Letters
A journal of the Society for Conservation Biology
Open Access

WILEY

Community management yields positive impacts for coastal fisheries resources and biodiversity conservation

Patrick F. Smallhorn-West[1,2,3] 🅾 | Karen Stone[4] | Daniela M. Ceccarelli[2] 🅾 |
Siola'a Malimali[5] | Tu'ikolongahau Halafihi[5] | Tom C. L. Bridge[2,6] 🅾 |
Robert L. Pressey[2] 🅾 | Geoffrey P. Jones[1,2] 🅾

[1] Marine Biology and Aquaculture, College of Science and Engineering, James Cook University, Townsville, QLD, Australia

[2] Australian Research Council Centre of Excellence for Coral Reef Studies, James Cook University, Townsville, QLD, Australia

[3] WorldFish, Jalan Batu Maung, Bayan Lepas, Penang, Malaysia

[4] Vava'u Environmental Protection Association, Neiafu, Vava'u, Tonga

[5] Ministry of Fisheries, Tongatapu, Nuku'alofa, Tonga

[6] Biodiversity and Geosciences Program, Museum of Tropical Queensland, Queensland Museum Network Townsville, QLD, Australia

Correspondence
Patrick Smallhorn-West
Email: patrick.smallhornwest@jcu.edu.au

Funding information
National Geographic Society, Grant/Award Number: CP-137ER-17; Australian Research Council Centre for Excellence in Coral Reef Studies: CE0561435

Abstract

Combining no-take marine reserves with exclusive access by communities to unreserved waters could provide the required incentives for community management to achieve positive impacts. However, few protected areas have been critically evaluated for their impact, which involves applying counterfactual thinking to predict conditions within protected areas if management had never occurred. Here, we use statistical matching to conduct a rigorous impact evaluation of dual management systems on coral reef fishes in Tonga, with communities having both full no-take areas and areas of exclusive fishing rights. No-take areas generally had positive impacts on the species richness, biomass, density, and size of target reef fish, while exclusive access areas were similar to predicted counterfactual conditions. The latter is likely because overall fishing pressure in exclusive access areas might not actually change, although more fish could be exploited by communities with access rights. Our findings suggest that dual management is effective at incentivizing effective community-based no-take areas for biodiversity conservation and resource management.

KEYWORDS

comanagement, conservation, locally managed marine area, marine protected areas, South Pacific, TURF

1 | INTRODUCTION

There is increasing evidence that appropriately situated marine protected areas (MPAs) with high compliance can produce positive outcomes for biodiversity and fisheries (Edgar et al., 2014; Gaines, White, Carr, & Palumbi, 2010). However, expansion of MPAs can be resisted by resource users over issues such as forced displacement, loss of access to seafood, and unfulfilled promises (Agardy, Notarbartolo, & Christie, 2011; Charles & Wilson, 2009). Balancing conservation priorities with human needs remains one of the key concerns in protected area research (Charles & Wilson, 2009).

Community-based marine management, whereby natural resource or biodiversity protection is conducted by, for, and with local communities (Western & Wright, 1994) is seen as one of the best approaches to strike a balance between the interests of biodiversity conservation and resource users (Jupiter, Cohen, Weeks, Tawake, & Govan, 2014). However, despite widespread acceptance of community-based or comanagement approaches, there are concerns that their expansion is driven by livelihoods and well-being objectives while benefits to biodiversity conservation are limited (Bartlett, Pakoa, & Manua, 2009). Therefore, even if positive ecological impacts are achieved locally as cobenefits with socially focused objectives, they might not scale to reach national or international biodiversity objectives (Gaines et al., 2010). Furthermore, if local priorities conflict with broader goals, then allowing resource users to take over management could result in prioritization of immediate benefits at the expense of long-term national or international objectives, such as biodiversity conservation or sustainable development.

In order for community management to achieve both local and national or international objectives, it is critical to identify incentives for local actions to ensure long-term change at a broad scale (Brockington & Schmidt-Soltau, 2004; Ferraro & Hanauer, 2011). Access restrictions, such as found within territorial use rights for fisheries (TURFs) or locally managed marine areas (LMMAs), are fisheries management tools by which communities or groups of fishers are given distributed or inherited access rights to a portion of the ocean (Gelcich et al., 2012; Jupiter et al., 2014; Villaseñor-Derbez et al., 2019). Access restrictions can promote a sense of stewardship and incentivize communities to sustainably manage their resources (Gelcich et al., 2012). Importantly, access restrictions and no-take marine reserves are not mutually exclusive (Jupiter et al., 2014; Villaseñor-Derbez et al., 2019). Instead they can be combined, whereby access restrictions can act as the incentive for establishing no-take reserves when communities might not otherwise be willing to give up areas for conservation (Smallhorn-West et al., 2020b).

The effectiveness of community-based marine management should be assessed by its impact, defined as the intended or unintended consequences that are directly or indirectly caused by an intervention (Adams, Barnes, & Pressey, 2019; Pressey, Visconti, & Ferraro, 2015). However, determining impact can be challenging because it involves estimating the counterfactual condition if no action or a different action had been taken (Ferraro, 2009; Pressey, Weeks, & Gurney, 2017). Estimating counterfactuals requires quantifying the extent to which observed conditions are the result of the intervention, or whether environmental or social contextual factors are masking failure

or exaggerating success (Adams et al., 2019). While impact evaluation techniques are well developed in many other fields of research (e.g., medicine, education and development aid) (White, 2009), few established marine protected areas have been critically evaluated for their impact (but see Ahmadia et al., 2015; Cinner et al., 2018; Cinner et al., 2020; Gill et al., 2017; Smallhorn-West, Weeks, Gurney, & Pressey, 2020a). Here, we conduct a rigorous impact evaluation using statistical matching to determine the ecological impact of a dual approach to community-based marine management combining access restrictions and no-take reserves. We focus on Tonga's national Special Management Area (SMA) program, in which communities are granted exclusive access to fishing grounds (SMAs) in exchange for making parts of them permanent no-take zones. The no-take zones are locally called Fish Habitat Reserves (FHRs), the size and location of which are determined at the communities' discretion. While the local objectives are based largely on reviving coastal fisheries resources, Tonga is also committed to various international biodiversity conservation targets (e.g., the Convention for Biological Diversity Strategic Plan for Biodiversity 2011–2020 and the 20 Aichi targets) (Anon, 2013) and the SMA program is the primary focus of conservation efforts in the country. We conducted ecological surveys and analysis to compare the current ecological state of Tonga's oldest SMAs to their estimated counterfactual conditions to determine whether both SMAs and FHRs can yield positive impacts for coastal fisheries resources and biodiversity conservation.

2 | METHODS

Tonga's SMA program launched in 2006 and, as of October 2019, includes 93 SMA or FHR areas (Smallhorn-West et al., 2020b). Our impact evaluation covers only SMAs established prior to 2014 and at least 3 years old at the time of ecological surveys. These requirements applied to seven SMA communities (with corresponding FHRs) (Figure 1), which were spread across the three main island groups in Tonga, with two in Tongatapu, four in Ha'apai, and one in Vava'u.

Ecological surveys were conducted from 2016 to 2018 across 375 sites in Tonga, both inside and outside FHRs and SMAs (Table S1). Areas open to fishing and newly implemented management areas were classified as control areas, providing the pool of control transects that could then be matched with transects in managed areas. At each site, four to six 30-m belt transects were laid parallel to the reef contour at depths of 3–12 m, resulting in a minimum of 12 transects within each SMA and FHR. The abundance and size of all large mobile fish were recorded to species level
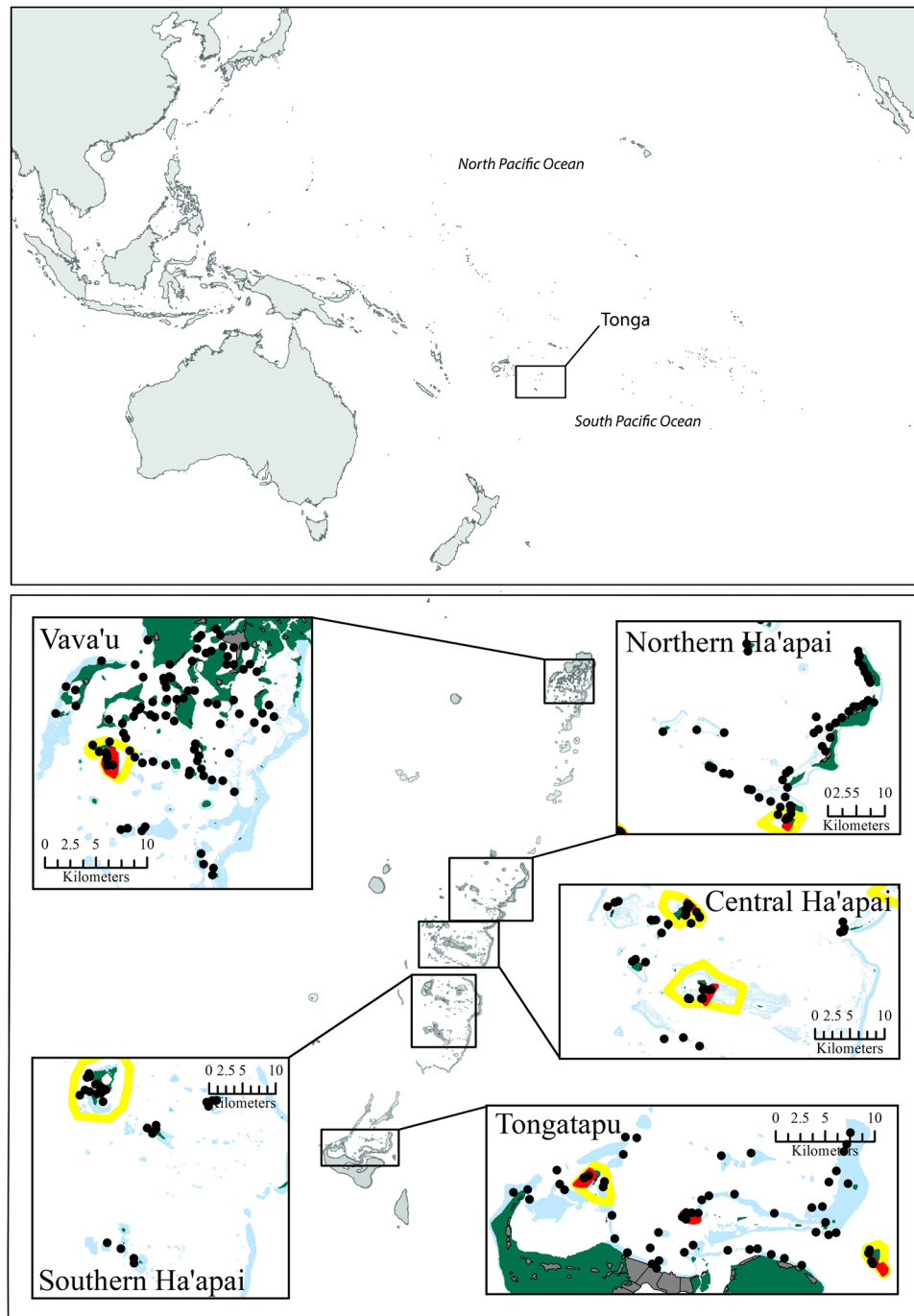
FIGURE 1 Map of Tonga showing the 14 Special Management Areas and Fish Habitat Reserves included in the impact evaluation. Yellow denotes Special Management Areas and red denotes no-take Fish Habitat Reserves. Black circles denote survey sites. Green represents land and grey indicate villages.

within a 5-m belt (Table S2). All small, site-attached reef fish species were recorded along a 2-m belt. The length and abundance of reef fish were converted to biomass following published length–weight relationships for each species (www.fishbase.org). Nineteen outcome variables of reef fish community composition were selected as meaningful indicators that aligned with the intended management objectives of the SMA program and international biodiver-

sity targets (Tonga Fisheries Division, Ministry of Agriculture & Food, 2010). These 19 outcome variables were: total reef fish species richness, total and family level biomass, density and mean total length of the five most commonly targeted reef fish families (Parks, 2017) (Acanthuridae, Lethrinidae, Lutjanidae, Scaridae, and Serranidae). We selected a 20 cm size cut off for biomass and density values because larger sized fish represent the fishable biomass

TABLE 1 Eleven contextual factors that were included in the matching model and used to estimate counterfactual conditions for transects inside Fish Habitat Reserves and Special Management Areas

Variable	Description	Reference
Depth	Depth (m), collected *in situ*.	Lindfield, McIlwain, and Harvey (2014)
Distance to land	Distance (m) from the nearest land source (Smallhorn-West et al., 2020).	Cinner, Graham, Huchery, and MacNeil (2013)
Distance to village	Distance (m) from the closest village (Smallhorn-West et al., 2020).	Cinner et al. (2013)
Fishing pressure	Normalized (0–100) abundance of commercial and subsistence fishers (adjusted for catch) extrapolated across the coral reefs of Tonga. It constitutes a unit-less value of relative long-term fishing effort throughout the region (Smallhorn-West et al., 2020).	Wilson et al. (2010)
Habitat	Exposed, semiexposed, or fringing, collected *in situ*.	Wilson et al. (2010)
Island group	Ha'apai, Tongatapu, or Vava'u.	-
Total live coral cover (%)	Collected either by the point intercept method or from photo quadrats annotated using the automated image analysis software CoralNet and BenthoBox.	Wilson et al. (2010)
Habitat macrocomplexity	Estimate of habitat complexity collected *in situ* on a five-point scale from low and sparse relief (score = 1) to exceptionally complex with numerous caves and overhangs (score = 5).	Wilson et al. (2010)
Slope	Estimate of reef slope collected *in situ* on a five-point scale from <10° (score = 1) to 90° (score = 5).	Ceccarelli (2016)
Surveyor	One of four surveyors	-
Wave energy	Average daily wave energy (joules per m^2) (Smallhorn-West et al., 2020).	Mumby et al. (2013)

of target reef fish species currently available to fishers and likely to be targeted.

We then selected 11 contextual factors to use for statistical matching. These encompassed environmental and social features of coral reefs that are known to influence either the response variables or the configuration of protected areas (Table 1). Details of the methodology behind these variables are available in Smallhorn-West et al. (2020).

2.1 | Impact evaluation

Counterfactual predictions for managed areas were estimated by statistically matching SMA and FHR transects to a large pool of control transects according to the characteristics of their covariates, using a combination of fixed and propensity score matching (Stuart, King, Imai, & Ho, 2011; R Core team, 2017). Propensity scores summarize statistically many covariates into a single score (Olmos & Govindasamy, 2015). They are defined as the conditional probability of assigning a unit to a particular treatment (i.e., likelihood of management as SMA or FHR), given a set of observed covariates ($z = i|X$), where z = treatment, i = treatment condition, and X = covariates. The probability of assignment is estimated using a logistic regression model, where treatment assignment is regressed on the set of observed covariates. The propensity score then allows

matching of transects with the same likelihood of receiving management.

Matching was conducted at the transect level. FHR and SMA transects were analyzed separately, but matched to the same overall pool of control transects (Figure 2) (Tables S3 and S4). The variables habitat type, island group, and surveyor were all fixed so that control transects could be paired only with managed transects if they matched the exact combination of these covariates. Following fixed matching, all remaining covariates were weighted equally, and the nearest neighbor distance was used to match transects with the closest propensity score first. We sampled with replacement, meaning control transects could be matched with multiple managed transects. In addition, each managed transect could also be paired with multiple control transects and, if multiple matches occurred, the mean was used as the estimated counterfactual. A prespecified tolerance (i.e., caliper) of 0.25 standard deviations of the sample estimated propensity scores was set to ensure only high-quality matches (Olmos & Govindasamy, 2015).

Covariate balance (i.e., the difference in the distribution of covariates across managed and control transects) was tested prior to and following matching by estimating the normalized difference between managed and control transects for each covariate in the model. An omnibus test, which tests whether at least one variable in the model is unbalanced, was conducted using the XBalance routine
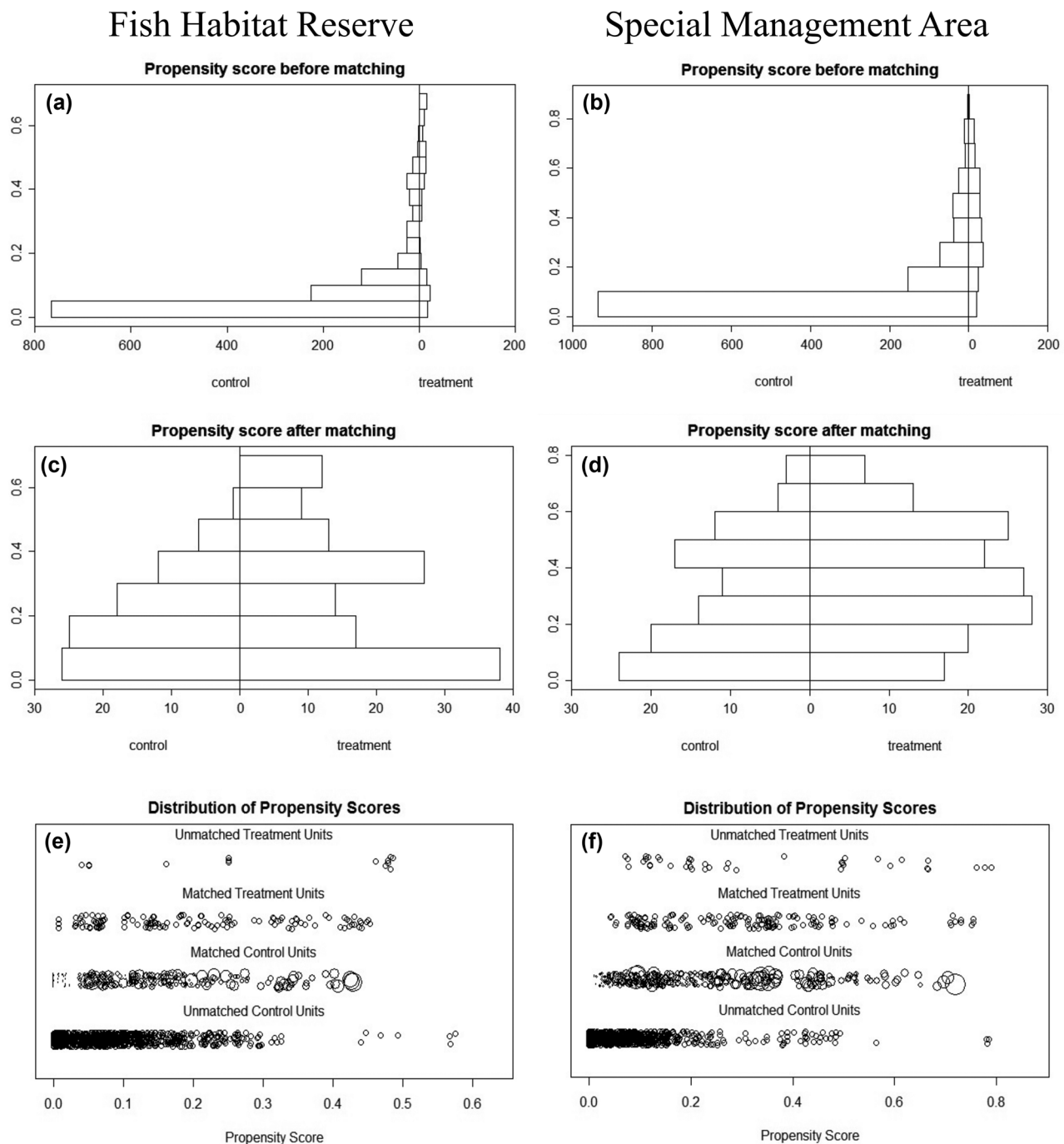
Fish Habitat Reserve

Special Management Area

FIGURE 2 Details of the matching procedure. Parts (a) and (c) represent frequency histograms of propensity scores prior to and following matching for Fish Habitat Reserves. Parts (b) and (d) represent frequency histograms of propensity scores prior to and following matching for Special Management Areas. Parts (e) and (f) show jitter plots of the propensity score distributions of matched and unmatched transects following matching for Fish Habitat Reserves and Special Management Areas, respectively. The size of the each control circle represents the number of treatment transects with which it was matched. All unmatched transects were discarded from the analysis

via a chi-squared test (Stuart, King, Imai, & Ho, 2011). For standardized differences, values over 25% between managed and control transects are considered unbalanced (Olmos & Govindasamy, 2015). Following matching there was no evidence of imbalance for FHRs or SMAs (Figure 2)

(Table S5). A total of 129 out of 143 FHR transects and 159 out of 200 SMA transects were matched to 247 and 397 control transects, respectively. All remaining unmatched managed and control transects were discarded from the analysis.

Finally, linear mixed effect models with community and site included as random factors, with site nested within community, were used to test the overall differences between matched FHR or SMA and control areas across each of the 19 outcome variables. Models were created with both fixed and random slopes and the one with the lowest AIC score selected. All biomass and density variables were $\log(x + 1)$ transformed. Model fit was examined using partial residual plots and tested with chi-squared tests on the residual sum of squares and residual degrees of freedom.

3 | RESULTS

Overall, there were consistent positive ecological impacts of FHRs (Figure 3, Table 2). Both overall target species biomass and density were approximately 5.3 and 3.6 times greater, and species richness 15% higher, inside no-take reserves than matched control transects. These impacts were most pronounced in the Scaridae family, with 3.7 times and 2.5 times as much biomass and density of scarids inside FHRs than in control transects, respectively. Although the overall density of Lethrinidae and Lutjanidae were small compared to other families, FHRs still supported 70% greater densities than control transects. Fish were also on average larger inside FHRs, with the mean total length of four of the five main target reef fish families 2–6 cm greater inside FHRs than matched control transects.

There was limited evidence of ecological impacts inside the SMAs. The most consistent trend was a small increase in the average size of the five main target reef fish families inside SMAs, although this was significant at $p < .05$ only for Lethrinidae and Scaridae. This trend was not evident in the biomass or density of target reef fish and, in three instances, biomass and density were significantly lower inside SMAs than in control sites (Lutjanid biomass and density and Serranid biomass). There was no evidence of an SMA effect on overall reef fish species richness.

4 | DISCUSSION

This study demonstrates that the dual approach to community-based marine management in Tonga, including exclusive access areas and associated no-take reserves, can be scaled up to achieve meaningful impacts for both coastal fisheries resources and biodiversity conservation. The success of the no-take areas is likely linked to the incentive provided by exclusive access to and greater control over local resources. While there were few quantifiable impacts of exclusive access areas, overall the combination of having both types of management areas is positive. Our

study provides one of the first full impact evaluations of a country's MPA network that has incorporated counterfactual analyses and is quantifiably robust to contextual conditions (but see Gill et al., 2017). This approach can therefore be used as a template by which to structure future impact evaluations of MPAs. In addition, Smallhorn-West, Bridge, Malimali, Pressey, and Jones (2019) also provide detailed recommendations for key conditions that should be in place for this approach to be useful. These results have important implications for management of reefs and for understanding how to balance the competing goals of improving coastal fisheries resources and biodiversity conservation in developing nations.

At the outset, it is important to acknowledge that, while positive ecological impacts are evident within the no-take FHRs, these represent only a small fraction (3% as of October 2019), of Tonga's total coral reef area. Fish stocks and species richness will likely continue to increase within Tonga's network of FHRs as new areas are implemented and existing areas grow older. However, despite these improvements, it is unclear whether the FHRs and any potential spillover will be sufficient to meet food supply needs while maintaining coral reef ecosystem function. In addition, given the lack of visible ecological impacts within the SMAs, it remains unclear the extent to which these areas are changing patterns of food consumption and nutrition within Tongan communities. Therefore, given the objective of "reviving the health and status of coastal fisheries resources for current and future generations" (Fisheries Division, Ministry of Agriculture & Food, 2010), additional management actions, such as changing fishing practices for the inshore commercial fisheries (2015 Tonga Fisheries Sector Plan, Section 42), along with the continued expansion of the SMA program, might be necessary to achieve broad objectives. Management practices such as SMAs therefore represent a platform from which to build in order to make additional progress toward many of the national and international biodiversity and sustainability targets.

Our findings can be used to inform policy actions for the expansion of local marine management by governments, researchers, and NGOs. First, the lessons gleaned from the SMA program and this study in particular provide policy-relevant evidence for other countries interested in expanding community-based marine management, such as the importance of localized incentives. Second, within Tonga these results also help improve the understanding of Tonga's progress toward achieving national and international targets for marine biodiversity conservation and sustainability. Table S6 outlines the relevance of our findings to both national (e.g., the 2015 Tonga Fisheries Sector Plan and the National Strategic Biodiversity Action Plan) and international (e.g., the Aichi 2020 Biodiversity targets)
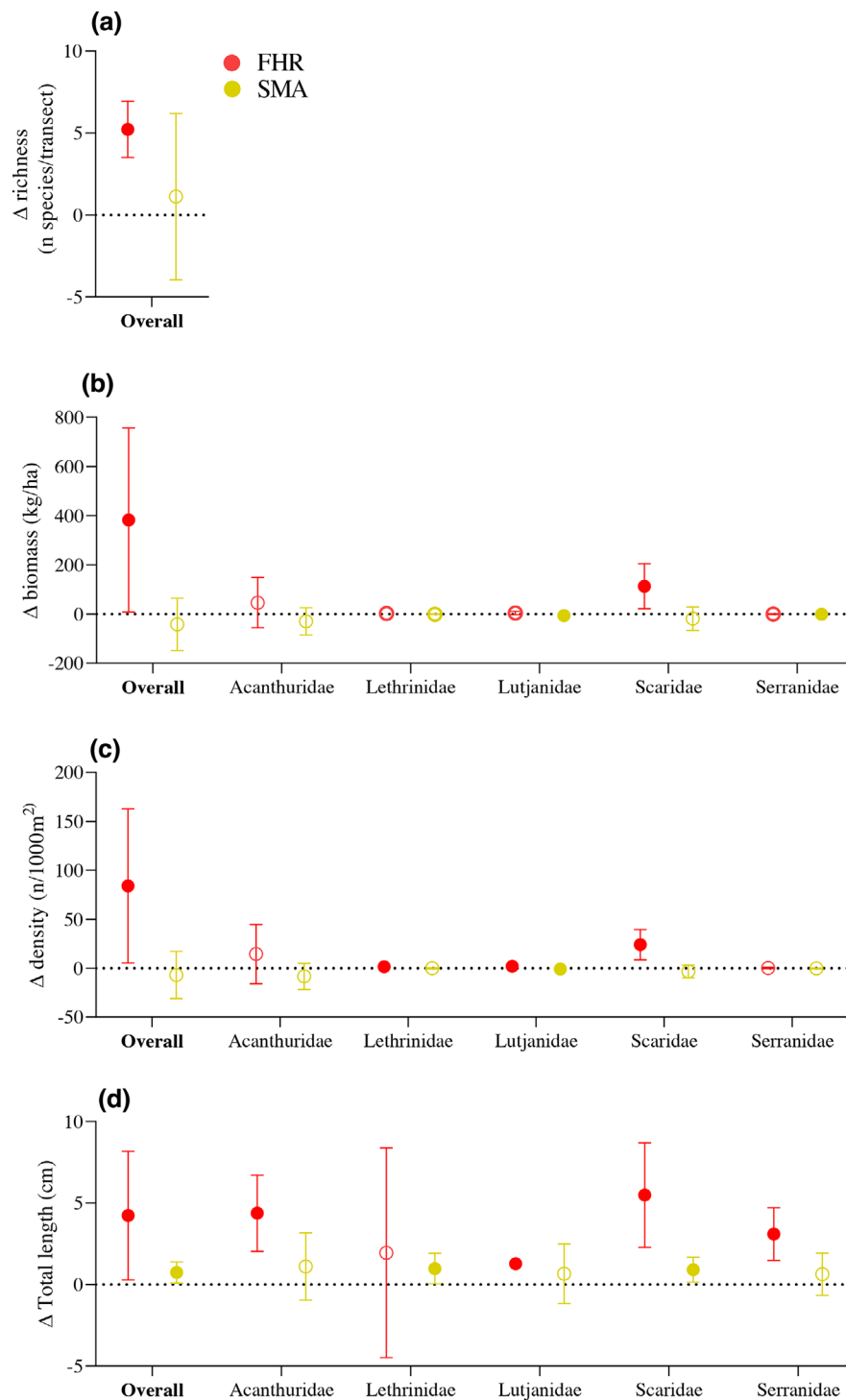
FIGURE 3 Ecological impacts of Tonga's Special Management Area program plotted as the mean difference between matched Fish Habitat Reserve or Special Management Area transects and control transects with ±95% confidence intervals. Closed circles represent values with margins not overlapping zero and statistically significant to $p < .05$. (a) Total reef fish species richness; (b) biomass of target species (>20 cm total length); (c) density of target species (>20 cm total length); and (d) mean total length of target species (juvenile to adult). Biomass and density plots represent differences in sum totals between transects and therefore overall values are cumulative of each family. The total length plot signifies differences in mean size of individual fish and therefore the overall columns represents the mean difference across all families

TABLE 2 Model results for mixed effect models examining the ecological impacts of Tonga's Special Management Area program shown as absolute mean ±95% confidence interval values of matched Fish Habitat Reserve or Special Management Area and control transects

| Family | Variable | Fish habitat reserve | | | | | | | | |
		Treatment	LCL	UCL	Control mean	LCL	UCL	df	t Score	p Value
Overall	Richness	39.24	35.29	43.18	34.02	30.07	37.96	196	6.019	<.05
	Biomass	469.88	83.52	856.24	87.94	−30.19	206.07	6	2.499	<.05
	Density	116.46	29.53	203.39	32.43	0.55	64.31	6	2.613	<.05
	Total length	22.41	19.50	25.31	18.18	12.68	23.67	196	2.115	<.05
Acanthuridae	Biomass	77.56	−29.52	184.64	31.56	−18.22	81.33	6	1.105	.311
	Density	26.96	−5.66	59.58	12.60	−5.13	30.34	6	1.155	.292
	Total length	16.13	13.36	18.89	11.75	8.38	15.13	196	3.693	<.05
Lethrinidae	Biomass	5.64	1.22	10.05	3.55	0.77	6.34	6	1.823	.118
	Density	3.27	1.81	4.74	1.95	1.08	2.82	6	2.972	<.05
	Total length	21.50	19.10	23.91	19.56	10.97	28.15	196	0.596	0552
Lutjanidae	Biomass	11.78	4.12	19.44	8.15	2.85	13.45	6	1.273	.250
	Density	4.60	2.86	6.33	2.69	1.68	3.71	6	2.976	<.05
	Total length	29.38	23.67	36.48	23.01	16.89	31.34	196	2.193	<.05
Scaridae	Biomass	155.46	38.64	272.29	42.46	10.55	74.38	6	3.017	<.05
	Density	39.48	17.41	61.55	15.53	6.85	24.20	6	3.760	<.05
	Total length	22.89	19.90	25.88	17.40	12.90	21.90	196	3.382	<.05
Serranidae	Biomass	2.99	0.98	4.99	3.27	1.07	5.46	6	−0.463	.660
	Density	2.01	1.26	2.76	1.91	1.20	2.63	6	0.431	.681
	Total length	21.23	17.48	24.99	18.14	14.38	21.89	196	3.805	<.05

| Family | Variable | Special management area | | | | | | | | |
		Treatment	LCL	UCL	Control mean	LCL	UCL	df	t Score	p Value
Overall	Richness	34.89	29.23	40.56	33.77	28.99	38.55	274	0.436	.663
	Biomass	143.54	13.30	273.78	185.64	3.73	367.54	6	−0.973	.368
	Density	47.26	8.96	85.56	54.27	18.46	90.07	6	−0.711	.504
	Total length	21.07	18.34	23.80	20.33	17.60	23.06	274	2.228	<.05
Acanthuridae	Biomass	19.61	−7.12	46.34	49.61	−22.53	121.75	6	−1.322	.234
	Density	9.68	−2.18	21.53	18.09	−1.89	38.07	6	−1.542	.174
	Total length	14.25	11.93	16.56	13.14	9.05	17.23	274	1.055	.292
Lethrinidae	Biomass	2.49	1.51	3.46	3.62	2.20	5.04	6	−2.152	.075
	Density	1.83	1.41	2.26	1.99	1.53	2.46	6	−0.773	.469
	Total length	19.73	17.95	21.50	18.74	16.97	20.52	274	2.076	<.05
Lutjanidae	Biomass	2.65	0.74	4.56	9.29	2.59	16.00	6	−3.085	<.05
	Density	1.93	1.35	2.50	2.95	2.07	3.83	6	−3.745	<.05
	Total length	29.36	25.67	33.05	28.70	25.01	32.39	274	0.710	.478

(Continues)

TABLE 2 (Continued)

Family	Variable	Special management area			Control mean	LCL	UCL	df	t Score	p Value
		Treatment	LCL	UCL						
Scaridae	Biomass	82.44	20.98	143.91	101.38	43.83	158.94	6	−0.981	.364
	Density	25.64	14.37	36.91	29.07	16.29	41.85	6	−1.288	.245
	Total length	20.91	17.52	24.29	19.99	16.61	23.38	274	2.373	<.05
Serranidae	Biomass	2.12	0.52	3.72	3.70	0.91	6.48	6	−2.517	<.05
	Density	1.67	1.11	2.22	2.01	1.34	2.68	6	−2.068	.084
	Total length	22.11	17.46	26.77	21.48	16.83	26.13	274	0.969	.333

LCL = lower confidence limit; UCL = upper confidence limit.

Controls represent the mean of all matched transects. Biomass is measured as kilograms per hectare of target species (> 20 cm total length). Density is measured as the number of individuals per 1,000 m^2 of target species (> 20 cm total length). Species richness is measured as the number of reef fish species per transect. Length is measured as total length in centimeters.

objectives. For example, improvements to reef fish density, biomass, and size provide support that the SMA program promotes environmentally sound practices for the management of marine resources (e.g., National Strategic Biodiversity Action Plan section 2.3). This study also provides evidence that the SMA program supports progress toward international targets such as Aichi targets 6, 10, and 11. However, while progress toward national and international targets is being made, caution is needed when using these targets to quantify success. For example, there is concern that too much focus on the area-based Aichi target 11, which aims to protect 10% of marine area by 2020, is encouraging minimal overlap between pressures and protection by favoring large, offshore reserves (Devillers et al., 2015). Care should therefore be taken in considering the conservation impacts of management, regardless of contributions to area protected or even representativeness (Pressey et al., 2017). Despite these caveats, it is clear that Tonga's SMA program represents a strong positive step for the country toward improving marine sustainability and biodiversity conservation.

A key principle in MPA design is that the size of no-take MPAs should be sufficient to incorporate the home ranges of the species they are intended to protect (Weeks, Green, Joseph, Peterson, & Terk, 2016). Numerous studies have also demonstrated that larger no-take MPAs are more likely to achieve positive results than smaller reserves (e.g., Edgar et al., 2014). However, the largest of Tonga's FHRs is only 2.6 km^2, and many are less than 1 km^2; yet they still consistently result in positive impacts, albeit across a limited total extent. While counterintuitive, these findings are consistent with other studies demonstrating that even small reserves (< 1 km^2) can produce significant biological responses (Bonaldo, Pires, Roberto, Hoey, & Hay, 2017; Russ & Alcala, 1996; Russ, Alcala, Maypa, Calumpong, & White, 2004). Given that the home ranges of many key tar-

get species are larger than the areas set aside for management, it is unclear how fishes are avoiding capture if they move beyond the boundaries of the FHR. Many protected areas globally are less than 1 km^2 in size (Costello & Ballantine, 2015) and further studies are necessary to investigate this effect; but there is evidence that some fishes, even wide-ranging species, can alter their behavior within a short timeframe to maximize the protection offered by no-take zones (Mee, Otto, & Pauly, 2017). In addition, the observed differences in recovery between reef fish families could also be due to the faster growth rates of scarids (Grandcourt, 2002), combined with the relatively young age of the SMA program.

The results of this study provide little evidence for positive ecological impacts within SMA areas, where fishing still occurs. This result is consistent with a recent global meta-analysis of MPA effectiveness demonstrating that moderately protected MPAs rarely perform better than unprotected areas (Zupan et al., 2018). However, while FHRs were established to explicitly address conservation objectives, the goals of SMAs are primarily socioeconomic. Key management objectives for SMAs are to "raise community awareness on fisheries conservation and management, promote sustainable fishing practices and improve living standards within the community" (Fisheries Division, Ministry of Agriculture & Food, 2010). In addition, SMAs are generally seen as a way to reestablish customary tenure, which is common in many Pacific nations but was lost in Tonga, and to prevent large-scale commercial fishing activities from destroying local food security (Gillett, 2017). As such SMAs might still be achieving their desired objectives even if there is no observed ecological change. Furthermore, any recovery of target species likely to occur from reduced fishing pressure by sources outside the community could be counteracted by increased local fishing. That in most cases ecological impacts inside SMAs were

not negative suggests that exclusive access is not increasing net fishing pressure, merely changing who fishes (Polunin, 1984), and therefore the net benefit of the dual system is positive. Ultimately the impacts of SMAs are be more likely to be found in people's nutrition and in their understanding of marine management than in the ecosystem itself.

Jupiter et al. (2014) outlined six management actions to achieve a broad range of objectives in community-based marine management: permanent closures, periodically harvested closures, species restrictions, gear restrictions, access restrictions, and alternative livelihood strategies. Within this framework, Tonga's SMA program represents a combination of access restrictions (i.e., SMAs) and permanent closures (i.e., FHRs). A key drawback suggested for access restrictions is that they might not be sufficient to maintain biomass or enhance sustainability, and that they "will not necessarily change the volume harvested, just who harvests it" (Jupiter et al., 2014; Polunin, 1984). However, these authors also suggested that access restrictions might be necessary to facilitate other management actions. While these other actions, such as permanent closures, might have strong evidence to support their effectiveness, there was concern that, given they are not historically prevalent in the Pacific (Johannes, 1978), there could be social barriers to their effective implementation (Foale & Manele, 2004). Tonga's management program builds on this hypothesis by utilizing SMAs (i.e., access restrictions) as necessary tools, despite no evident ecological impacts, to incentivize the implementation of FHRs (i.e., permanent closures). Given the open access history of Tonga's marine management (Gillett, 2017), FHRs might have had little support otherwise.

Tonga's SMA program represents a successfully vetted combination of management actions to add to the tool kit of marine managers aiming to achieve ecological impact in the community context. However, the success of this program has relied on reinventing customary tenure in a country with little historical management. While this approach has been successful in Tonga, other countries with stronger traditional access rights might have greater difficulty in providing incentives for permanent closures. A key consideration is therefore that support for this program will likely be greatest in areas where previous management is weakest. Determining the historical context of community priorities and using these to successfully incentivize conservation will be a key factor in the successful implementation of this framework in other regions.

AUTHOR CONTRIBUTION

PSW conceived and wrote the manuscript and analyzed the data. KS and DMC conducted fieldwork and provided data. SM and TH contributed to in country support and critical background information about the program. TB, RP, and GJ supervised the project and developed the ideas. All authors contributed to the final manuscript.

ETHICS STATEMENT

All research activities were conducted in accordance with James Cook University Animal Ethic Guidelines (permit approval A2454).

DATA STATEMENT

Raw data is available on the data repository PANGAEA (https://doi.pangaea.de/10.1594/PANGAEA.917698).

CONFLICT OF INTEREST

The authors declare no conflicts of interest.

ORCID

Patrick F. Smallhorn-West https://orcid.org/0000-0001-6782-3704
Daniela M. Ceccarelli https://orcid.org/0000-0001-8067-8716
Tom C. L. Bridge https://orcid.org/0000-0003-3951-284X
Robert L. Pressey https://orcid.org/0000-0002-6531-7389
Geoffrey P. Jones https://orcid.org/0000-0002-6244-1245

REFERENCES

Adams, V. M., Barnes, M., & Pressey, R. L. (2019). Shortfalls in conservation evidence: Moving from ecological effects of interventions to policy evaluation. *One Earth*, *1*, 62–75. https://doi.org/10.1016/j.oneear.2019.08.017

Agardy, T., Notarbartolo, G., & Christie, P. (2011). Mind the gap: Addressing the shortcomings of marine protected areas through large scale marine spatial planning. *Marine Policy*, *35*, 226–232. https://doi.org/10.1016/j.marpol.2010.10.006

Ahmadia, G. N., Glew, L., Provost, M., Gill, D., Hidayat, N. I., Mangubhai, S., & Purwanto, Fox, H. E. (2015). Integrating impact evaluation in the design and implementation of monitoring marine protected areas. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *370*(1681), 20140275. https://doi.org/10.1098/rstb.2014.0275

Anon. (2013). CBD, Quick guides to the Aichi Biodiversity Targets. Version 2. 02/2013.

Bartlett, C. Y., Pakoa, K., & Manua, C. (2009). Marine reserve phenomenon in the Pacific Islands. *Marine Policy*, *33*(4), 673–678. https://doi.org/10.1016/j.marpol.2009.01.004

Bonaldo, R. M., Pires, M. M., Roberto, P., Hoey, S., & Hay, M. E. (2017). Small marine protected areas in Fiji provide refuge for reef fish assemblages, feeding groups, and corals, 12(1), e170638. https://doi.org/10.1371/journal.pone.0170638

Brockington, D., & Schmidt-Soltau, K. (2004). The social and environmental impacts of wilderness and development Oryx, *38*(2), 140–142. https://doi.org/10.1017/S0030605304000250

Ceccarelli, D. (2016). Vava'u SMA project baseline survey technical report. 1–74.

Charles, A., & Wilson, L. (2009). Human dimensions of marine protected areas. *ICES Journal of Marine Science*, *66*(1), 6–15.

Cinner, J. E., Graham, N. A. J., Huchery, C., & MacNeil, M. A. (2013). Global effects of local human population density and distance to markets on the condition of coral reef fisheries. *Conservation Biology*, *27*, 453–458.

Cinner, J. E., Maire, E., Huchery, C., MacNeil, M. A., Graham, N. A., Mora, C., … & D'agata, S. (2018). Gravity of human impacts mediates coral reef conservation gains. *Proceedings of the National Academy of Sciences*, *115*(27), E6116-E6125.

Cinner, J. E., Zamborain-Mason, J., Gurney, G. G., Graham, N. A., MacNeil, M. A., Hoey, A. S., … & Maina, J. M. (2020). Meeting fisheries, ecosystem function, and biodiversity goals in a human-dominated world. *Science*, *368*(6488), 307-311.

Costello, M., & Ballantine, B. (2015). Biodiversity conservation should focus on no-take Marine Reserves. *Trends Ecology Evolution*, 30, 507–509. doi: https://doi.org/10.1016/j.tree.2015.06.011 PMID: 26321055

Devillers, R., Pressey, R. L., Grech, A., Kittinger, J. N., Edgar, G. J., Ward, T., & Watson, R. (2015). 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

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

Ferraro, P. J. (2009). Counterfactual thinking and impact evaluation in environmental policy. *New Directions for Evaluation*, *2009*(122), 75–84. https://doi.org/10.1002/ev.297

Ferraro, P. J., & Hanauer, M. M. (2011). Protecting ecosystems and alleviating poverty with parks and reserves : 'Win-win' or tradeoffs. *Environmental and Resource Economics*, *48*(2), 269–286. https://doi.org/10.1007/s10640-010-9408-z

Fisheries Division Ministry of Agriculture & Food, Forests and Fisheries. (2010). Community-managed special management areas in Tonga Brochure, 1–2. Retrieved from http://www.tongafish.gov.to/images/documents/Publications/Brochures/SMAbrochure-tonga-En.pdf

Foale, S., & Manele, B. (2004). Social and political barriers to the use of marine protected areas for conservation and fishery management in Melanesia. *Asia Pacific Viewpoint*, *45*, 373–386.

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*, *107*, 18286–18293. https://doi.org/10.1073/pnas.0906473107

Gelcich, S., Fernández, M., Godoy, N., Canepa, A., Prado, L., & Castilla, J. C. (2012). Territorial user rights for fisheries as ancillary instruments for marine coastal conservation in Chile. *Conservation Biology*, *26*(6), 1005–1015.

Gillett, R. (2017). *A review of special management areas in Tonga* (Vol. FAO Fisher).

Gill, D. A., Mascia, M. B., Ahmadia, G. N., Glew, L., Lester, S. E., Barnes, M., … Holst, S. (2017). Capacity shortfalls hinder the performance of marine protected area globally. *Nature*, *543*(7647), 665.

Grandcourt, E. M. (2002). Demographic characteristics of a selection of exploited reef fish from the Seychelles: Preliminary study. *Marine and Freshwater Research*, *53*(2), 123–130.

Johannes, R. E. (1978). Traditional marine conservation methods in Oceania and their demise. *Annual Review of Ecology and Systematics*, *9*(1), 349–364.

Jupiter, S. D., Cohen, P. J., Weeks, R., Tawake, A., & Govan, H. (2014). Locally-managed marine areas: Multiple objectives and diverse strategies. *Pacific Conservation Biology*, *20*(2), 165–179.

Kingdom of Tonga's Fifth National Report to the Convention on Biological Diversity. (2014). Nuku'alofa. 197 pp.

Kingdom of Tonga National Biodiversity Strategy and Action Plan (2006). Department of Environment. 97 pp.

Lindfield, S. J., McIlwain, J. L., & Harvey, E. S. (2014). Depth refuge and the impacts of SCUBA spearfishing on coral reef fishes. *PloS One*, *9*(3), e92628.

Mee, J. A., Otto, S. P., & Pauly, D. (2017). Evolution of movement rate increases the effectiveness of marine reserves for the conservation of pelagic fishes. *Evolutionary Applications*, *10*(5), 444–461.

Mumby, P. J., Bejarano, S., Golbuu, Y., Steneck, R. S., Arnold, S. N., Van Woesik, R., & Friedlander, A. M. (2013). Empirical relationships among resilience indicators on Micronesian reefs. *Coral Reefs*, *32*(1), 213–226.

Olmos, A., & Govindasamy, P. (2015). Propensity scores : A practical introduction using R. *Journal of Multidisciplinary Evaluation*, *11*(25), 68–88.

Parks, J. (2017). Baseline socioeconomic survey of the Vava'u special management areas. As a component of the Tonga climate resilience sector.

Polunin, N. V. C. (1984). Do traditional marine "reserves" conserve? A view of Indonesian and New Guinean evidence. In *Traditional marine resource management in the Pacific Basin: An anthology* (pp. 267–283). K. Ruddle & R. E. Johannes (Eds.). Jakarta, Indonesia: UNESCO/ROSTSEA.

Pressey, R. L., Visconti, P., & Ferraro, P. J. (2015). Making parks make a difference: Poor alignment of policy, planning and management with protected-area impact, and ways forward. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *370*(1681). https://doi.org/10.1098/rstb.2014.0280

Pressey, R. L., Weeks, R., & Gurney, G. G. (2017). From displacement activities to evidence-informed decisions in conservation. *Biological Conservation*, *212*(July), 337–348. https://doi.org/10.1016/j.biocon.2017.06.009

R Core Team. (2017). *R: A language and environment for statistical computing*. Vienna, Austria: R Foundation for Statistical Computing. Retrieved from https://www.R-project.org/

Russ, G. R., & Alcala, A. C. (1996). Do marine reserves export adult fish biomass? Evidence from Apo Island, central Philippines. *Marine Ecology Progress Series*, *132*, 1–9.

Russ, G. R., Alcala, A. C., Maypa, A. P., Calumpong, H. P., & White, A. T. (2004). Marine reserve benefits local fisheries. *Ecological Applications*, *14*(2), 597–606.

Smallhorn-West, P. F., Bridge, T. C., Malimali, S. A., Pressey, R. L., & Jones, G. P. (2019). Predicting impact to assess the efficacy of community-based marine reserve design. *Conservation Letters*, *12*(1), e12602. https://doi.org/10.1111/conl.12602.

Smallhorn-West, P. F., Gordon, S. E., Dempsey, A. C., Purkis, S. J., Malimali, S., Halafihi, T., Southgate, P. C., Bridge, T. C. L., Pressey, R.L., & Jones, G. P. (2020). Tongan socio-environmental spatial layers for marine ecosystem management. *Pacific Conservation Biology*, https://doi.org/10.1071/PC19032.

Smallhorn-West, P. F., Weeks, R., Gurney, G., & Pressey, R. L. (2020a). Ecological and socioeconomic impacts of marine protected areas in the South Pacific: Assessing the evidence base. *Biodiversity and Conservation*, 1–32.

Smallhorn-West, P. F., Sheehan, J., Malimali, S., Halafihi, T., Bridge, T. C., Pressey, R. L., & Jones, G. P. (2020b). Incentivizing co-management for impact: mechanisms driving the successful national expansion of Tonga's Special Management Area program. *Conservation Letters*, e12742.

Stone, K., Mengerink, K., Estep, A., Halafihi, T., Malimali, S., & Matoto, L. (2017). Vava'u ocean initiative marine expedition interim report.

Stuart, E. A., King, G., Imai, K., & Ho, D. (2011). MatchIt: Nonparametric preprocessing for parametric causal inference. *Journal of Statistical Software*.

The Kingdom of Tonga: Tonga Fisheries Sector Plan. (2015). *Tonga ministry of fisheries*. Nuku'alofa 80 pp.

Villaseñor-Derbez, J. C., Aceves-Bueno, E., Fulton, S., Suarez, A., Hernandez-Velasco, A., Torre, J., & Micheli, F. (2019). An interdisciplinary evaluation of community-based TURF-reserves. *PloS One*, *14*(8), e0221660.

Weeks, R., Green, A. L., Joseph, E., Peterson, N., & Terk, E. (2016). Using reef fish movement to inform marine reserve design. *Journal of Applied Ecology*, *54*(1), 145–152. https://doi.org/10.1111/1365-2664.12736

Western, D., & Wright, R. M. (1994). *Natural connections: Perspectives in community-based conservation* (pp. 349–364). Washington, DC: Island Press.

White, H. (2009). Theory-based impact evaluation: Principles and practice. *Journal of Development Effectiveness*, *1*, 271–284. doi: https://doi.org/10.1080/19439340903114628

Wilson, S., Fisher, R., Pratchett, M., Graham, N., Dulvy, N., Turner, R., … Polunin, N. (2010). Habitat degradation and fishing effects on the size structure of coral reef fish communities.pdf. *Ecological Applications*, *20*(2), 442–451.

Zupan, M., Fragkopoulou, E., Claudet, J., Erzini, K., Horta e Costa, B., & Goncalves, E. J. (2018). Marine partially protected areas: Drivers of ecological effectiveness. *Frontiers in Ecology and the Environment*, *16*, 381–387.

SUPPORTING INFORMATION

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