REVIEW PAPER

Ecological and socioeconomic impacts of marine protected areas in the South Pacific: assessing the evidence base

Patrick F. Smallhorn-West, et al. *[full author details at the end of the article]*

Abstract

Marine protected areas (MPAs) in the South Pacific have a unique history that calls for a regional-scale synthesis of MPA impacts and the factors related to positive ecological and socioeconomic change. However, recommendations of best approaches to MPA implementation can be made only when evaluation techniques are sound. Impact evaluation involves quantifying the effects of an intervention over and above the counterfactual of no intervention or a different intervention. Determining the true impact of an MPA can be challenging because additional factors beyond the presence of an MPA can confound the observed results (e.g. differences in ecological or socioeconomic conditions between MPA and control sites). While impact evaluation techniques employing counterfactual thinking have been well developed in other fields, they have been embraced only slowly in the MPA evaluation literature. We conducted a structured literature search and synthesis of MPA evaluation studies from the South Pacific to determine: (i) the overall ecological and socioeconomic impacts of MPAs in the region, (ii) what factors were associated with positive, neutral, or negative impacts, and (iii) to what extent the MPA evaluation literature from the region has incorporated counterfactual thinking and robust impact evaluation techniques. Based on 52 identified studies, 42% of measured ecological impacts were positive. While 72% of socioeconomic impacts were positive, these were from only eight studies. The proportion of positive impacts was comparable between community-based and centrally governed MPAs, suggesting that both governance approaches are viable options in the region. No-take MPAs had a greater number of positive ecological impacts than periodic closures and there was little evidence of any long-term ecological recovery within periodic closures following harvesting. Importantly, more than half of the studies examined (59%) did not provide any clear consideration of factors beyond the presence of the MPA that might have confounded their results. We conclude that counterfactual thinking has yet to be fully embraced in impact evaluation studies in the region and recommend pathways by which progress can be made.

Introduction

Natural ecosystems are under increasing anthropogenic pressures, some of which can be mitigated by protected areas (Chape et al. 2005; Mora et al. 2006). Marine protected areas (MPAs) have a diversity of objectives that can include enhancing ecosystem resilience, protecting biodiversity, and benefiting fisheries livelihoods by fostering sustainable harvesting (Halpern and Warner 2003; Gaines et al. 2010; Mellin et al. 2016). As a result of international targets (i.e. Convention on Biological Diversity Aichi 11) calling for nations to protect 10% of coastal and marine areas by 2020 (Toonen et al. 2013; Thomas et al. 2014), MPAs are expanding globally. Because MPAs alter human behaviour in ecosystems, their impacts have both ecological and socioeconomic dimensions. The impacts of MPAs can therefore be broad and multifaceted, with perceived success relating to the specific objectives for which MPAs are established (Jupiter et al. 2014). These objectives can be large-scale, such as national commitments to protect biodiversity, or local, such as enhancing the food security of communities and ecosystem resilience, and a single MPA can be established to achieve multiple objectives.

At the outset, the term "impacts" of MPAs is problematic. Studies vary in the rigour with which they determine impact, and therefore in their reliability. The most rigorous technique involves formal impact evaluation (Ferraro and Hanauer 2014; Ferraro and Pressey 2015), with impact defined as the intended or unintended consequences that are directly or indirectly caused by an intervention (e.g. MPA implementation) (Table 1) (Mascia et al. 2014). Importantly, this definition of impact involves counterfactual analysis (Table 1), which supports causal inference by asking: what would have happened in the absence of the intervention (Pressey et al. 2015, 2017; Adams et al. 2019)? Determining impact in this rigorous way can be challenging because it involves identifying how much observed conditions are due to the intervention, and how much to confounding factors (Table 1) that can mask intervention failure or exaggerate success (Adams et al. 2019). For example, Andam et al. (2008) demonstrated that the actual impact of protected areas on deforestation in Costa Rica was confounded by most protected areas being located far from roads and in places that were unlikely to be deforested regardless of their status. Impact evaluation techniques employing counterfactual thinking have been well developed in fields other than conservation science, and although several studies have outlined quasi-experimental approaches for impact evaluation of protected areas (Ferraro and Hanauer 2014; Ahmadia et al. 2015), they have been embraced only slowly in the MPA evaluation literature (Pressey et al. 2017). Consequently, many studies aiming to estimate the impacts of MPAs have been limited by choice of counterfactual sites, often associated with little consideration of confounding factors. Despite this caveat, we refer throughout this paper to 'impacts' as estimates of MPA performance, acknowledging that these estimates vary in rigour and do not all constitute actual impact evaluationss. Part of our review assesses this variation in rigour.

The direct ecological impacts of MPAs are generally changes in biomass, abundance, and diversity of target species (Alcala 1988; Russ and Alcala 1996, 2004; Halpern and Warner 2003), which are associated with limiting acute disturbances such as fishing, destructive anchoring, or development. Indirect flow-on effects, such as changes in total

Table 1 Key terms and definitions

Term	Definition
Before-after (BA)[a]	An evaluation technique that measures outcome variable(s) prior to and following MPA(s) implementation. It assumes that there are no concurrent factors that may influence outcome variable(s) and therefore any changes are attributable to the MPA
Control-intervention (CI)[a]	An evaluation technique that measures outcome variable(s) at a single point in time at sites inside and outside MPA(s). It assumes that the outside site (control) accurately reflects the counterfactual condition of the MPA site. Specifically, it assumes that there are no differences between the control and MPA sites with respect to the outcome variables prior to the MPA being implemented and that the only factor that may influence outcome variables is the MPA
Before-after-control-intervention (BACI)[b]	An evaluation technique that measures outcome variable(s) at sites inside and outside of MPA(s) prior to and following MPA implementation. This technique relies on the parallel trends assumption, that, in the absence of management, changes in outcome variables in MPA sites would be the same as those in control sites
Before-after-control-intervention-paired-series (BACIPS)[c]	An evaluation technique that measures outcome variable(s) at paired sites inside and outside of MPA(s) prior to and following MPA implementation. This technique uses the average difference in the before period as a null hypothesis for the difference that would exist in the after period in the absence of an intervention. An addition to this approach is the progressive change BACIPS, which incorporates recovery rates into measurements of difference instead of assuming step-wise change following management
Matching[d]	Grouping MPA sites with one or more control sites based on statistical measurements of similarity across multiple ecological or socioeconomic factors. Matching can be incorporated into CI, BACI, and BACIPS experimental designs
Confounding factor	A known or unknown factor that can mask the true impact of an intervention, resulting in over or under-estimations of impact (see Table 3)
Counterfactual[e]	The outcome that would have occured in the absence of the intervention considered, or a different intervention
Factor	An element predicted to influence one or more reported variables. Factors can be those predicted to drive differences in MPA impacts (e.g. governance and management strategies), or those controlled for when selecting MPA and control sites (e.g. habitat and education level)
Impact[e]	The intended and unintended consequences (e.g. changes in knowledge and attitudes, behaviours, and/or social and environmental conditions) that are directly or indirectly caused by an intervention
Outcome[e]	The desired ends that interventions are intended to induce (e.g. changes in knowledge and attitudes, behaviours, achieved targets of fish abundance or coral cover)
Reflexive counterfactual (RC)[f]	Framing social perception questions in a way that attributes causality to the protected area (e.g. *Are there more fish because of the MPA?*) and uses the surveyed individuals' perceptions of pre-existing conditions as the comparator

Table 1 (continued)

Term	Definition
Variable	An indicator for which change is measured within a study (e.g. target species biomass, income, catch)

[a]Adams et al. (2019)

[b]Gertler et al. (2011)

[c]Thiault et al. (2019)

[d]Ahmadia et al. (2015)

[e]Ferraro (2009)

[f]Franks et al. (2014)

biodiversity, coral cover, or rates of herbivory, depend largely on changes in ecosystem dynamics based on the responses of target species (Mellin et al. 2016). Changes in ecological parameters can, in turn, influence socioeconomic impacts, such as fish catch and related income (Bartlett et al. 2009a; Mizrahi et al. 2018). While some of the socioeconomic impacts of MPA implementation derive from ecological impacts, others do not depend on changes in marine ecosystems. For example, direct socioeconomic impacts can include community empowerment (Egli et al. 2010) or conflict over unfairness in regard to management-related decision-making (Gurney et al. 2014).

An extensive body of literature has sought to understand factors (Table 1) related to MPA impact (e.g. Halpern 2003; Claudet et al. 2008; Lester et al. 2009; Vandeperre et al. 2011). Larger and older MPAs generally have more impact (Edgar et al. 2014), as well as those with adequate staff and budget capacity (Gill et al. 2016). Management practices, here defined as the rules by which access to a reserve is administered, can also differ, with potential benefits and limitations of different practices. For example, whilst is has been established that permanent no-take MPAs often have greater ecological impact than periodic closures (Edgar et al. 2014), in some instances conflicting interests between users have resulted in periodic closures being more effective at achieving direct ecological impact (Giakoumi et al. 2017; Goetze et al. 2017a). Likewise, different governance strategies, defined here as how authority for administration is allocated, also have their own strengths and weaknesses. Centralized governance of MPAs is common in high-income countries and typically focused on biodiversity conservation objectives, but might not incorporate local stakeholders' objectives, resulting in low support and compliance and, therefore, a reliance on enforcement (Gaymer et al. 2014). In contrast, community-based governance, which is more prevalent in countries with strong local tenure rights or where government resources are limited (Govan 2009b), often focuses on local objectives and can therefore have greater local support (Ostrom 1990; Cox et al. 2010), although broader conservation priorities might be achieved only incidentally (Ban et al. 2011). Importantly these strategies are not mutually exclusive and some countries have 'scaled-up' locally managed MPAs into broader networks (e.g. Fiji FLMMA) (Ban et al. 2011).

The body of work on MPA impacts varies widely in geographic scope, with both narrow and very broad scopes having limitations in identifying factors associated with positive impacts. Many studies have identified the impacts of individual or small groups of MPAs, and while these studies can demonstrate isolated successes and failures, they are unable to draw conclusions about different strategies within the same socio-economic

and political contexts. In contrast, global reviews of MPA impacts have often compared various approaches to MPA implementation (Lester et al. 2009; Selig and Bruno 2010; Edgar et al. 2014; Gill et al. 2016). However, the high inherent differences between MPAs in such a broad-scale approach (e.g. habitats, species, governance, funding, and enforcement) likely misses many regionally relevant factors associated with impact. Regional-scale analyses (e.g. Giakoumi et al. 2017; Kamil et al. 2017) are therefore useful because they are able to highlight factors that confer positive impacts from MPA implementation that have particular importance in those contexts and that might differ from global generalisations. In regional studies, perspectives on MPA impacts can, for example, be compared between countries with very different management strategies while controlling for similarities in habitat and governance.

Protected areas in the South Pacific have a unique history that calls for a region-specific and regional-scale synthesis of MPA impacts. The region has a long tradition of local marine management, arising from high population densities on small land areas, with a large dependence on marine resources (Johannes 1978; Govan 2009a). Western colonialism undermined traditional management with the imposition of new, centrally based laws by colonial powers and a breakdown of traditional authority (Johannes 1978). However, in the following 25 years, this process was sufficiently reversed for Johannes (2002) to retract his earlier appraisal, describing a renaissance of traditional marine management in Oceania. Even more recently, the commitment of Pacific Island nations to the CBD Aichi target 11 of 10% protected-area coverage for their marine and coastal waters has resulted in some countries creating additional large, centrally governed MPAs (e.g. Marae Moana—Cook Islands; Le Parc Naturel de la Mer de Corail—New Caledonia). This long and variable history, combined with strong local support and a rapid expansion of MPAs, has resulted in multiple management and governance strategies across a large area with relatively uniform habitat, culture, and environment. The South Pacific is therefore ideally suited to examine different factors associated with MPA impacts. However, despite the extensive MPA literature in the region, the extent to which MPA evaluators have embraced counterfactual thinking and robust impact evaluation techniques, including consideration of confounders, is still unclear.

In this paper, we conducted a structured literature search and synthesis of studies that have set out to estimate MPA impact from the South Pacific. Studies not able to demonstrate impact in the counterfactual sense often instead measure outcomes, defined as the desired ends that interventions are intended to induce (Mascia et al. 2014) (Table 1), and these studies are also included in this review. Our questions are divided into two sections. Part 1 asks: (i) what have been the overall ecological and socioeconomic impacts of MPAs in the South Pacific?, and (ii) what are the factors that have been associated with positive, neutral, or negative ecological or socioeconomic impacts? Part 2 questions to what extent the MPA impact literature from the South Pacific has embraced counterfactual thinking and robust impact evaluation techniques, including consideration of confounding factors. We conclude that there is room for improvement in how MPA evaluation studies are being conducted in the South Pacific.

Methods

Study region

This study chose a predefined search area based on what has been traditionally termed the South Pacific (Fig. 1). This is the region which most strongly identifies with the shifts

from traditional marine tenure to central colonialist management, back to the renaissance of community-based management, and to the current paradigm of large-scale MPA implementation aimed at achieving international CBD targets.

Literature search

A structured search and review of the MPA literature from the South Pacific was conducted in Google Scholar and Web of Science during January and February 2018, and in February 2019. The identification of studies, inclusion criteria for the review, and data extracted for analysis are summarized in Fig. 2. The search string was developed to include both locations and management terms specific to the region. Articles were screened by their title and abstract prior to full-text viewing based on pre-determined criteria. Articles were included for full-text viewing if they contained some sort of measurement of ecological or socioeconomic variables associated with the implementation, existence, or removal of an MPA in the South Pacific.

Impacts and related factors

For each study that satisfied the inclusion criteria (Fig. 2), the number and type (ecological or socioeconomic) of measured variables (Table 1) were recorded. All variables that were measured against a temporal or spatial control were examined and their difference (i.e. negative, no change, positive) from the control was noted. Rather than a traditional meta-analysis, which considers the relative effectiveness of each study at achieving a specific objective, we examined all outcome variables reported in each study to provide a comprehensive assessment of the impacts of MPA implementation relative to various factors. MPAs then
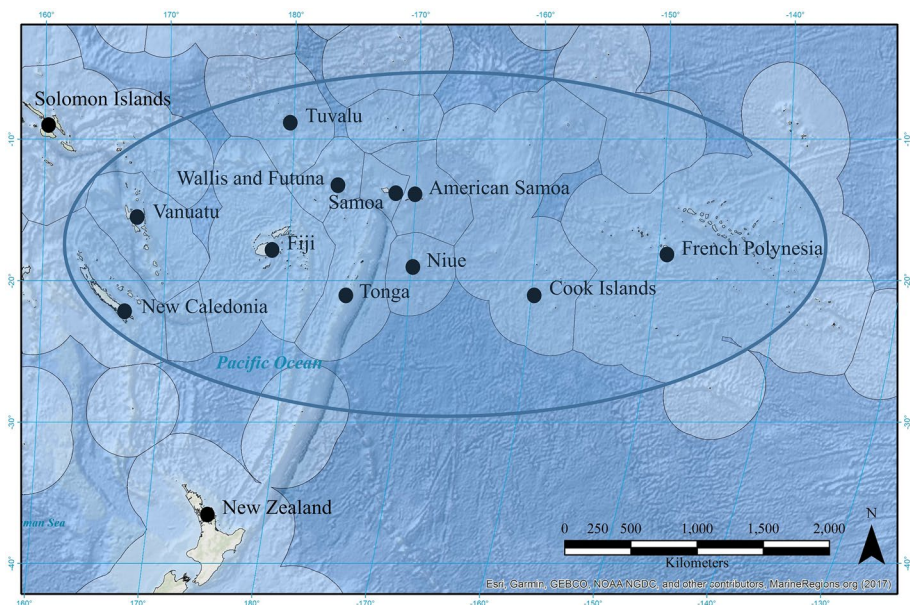

Fig. 1 The South Pacific, as defined as the study region for this review
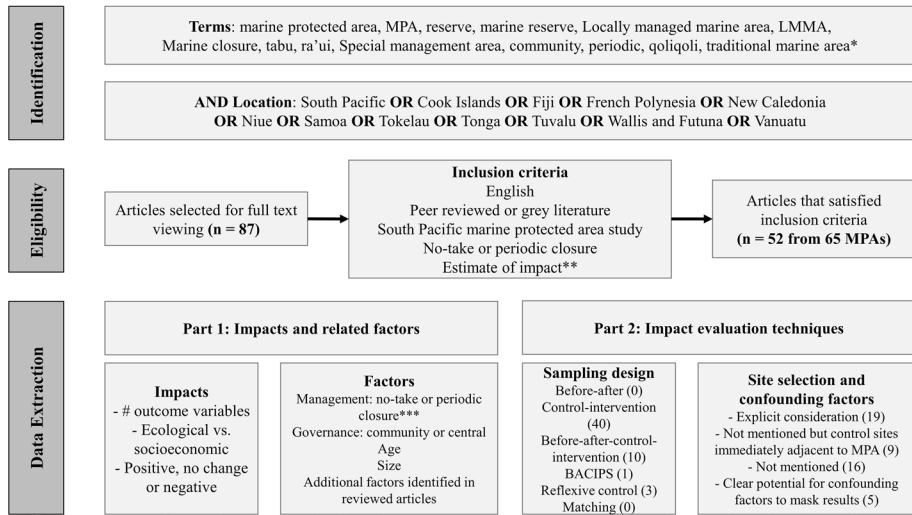
Fig. 2 Flow diagram for article screening and inclusion in the review. For part 2 of data extraction (Impact evaluation techniques), the number of studies utilizing different sampling methods and site selection criteria are included in brackets. *While not all the marine conservation interventions listed in the terms box are necessarily MPAs, these terms were nonetheless used for initial screening purposes. Once articles were selected for full text viewing, only interventions that incorporated fully closed or periodically harvested closures were included in the analysis. **Estimates of impact included any study using BA, CI, BACI, BACIPS, reflexive counterfactual or matching techniques (see Table 1 for definitions). While not all of these techniques necessarily quantify impact reliably, they were nonetheless included to assess how well studies in the region incorporated impact evaluation techniques. ***Impacts for periodic closures were measured at multiple time points: pre-harvest, immediately post-harvest, and following a recovery period

were categorised according to the factors of governance approach (i.e. central or community governed) and management strategy (i.e. no-take or periodically harvested), and size and age if data were available. Not all MPAs fitted neatly within these categories. For example, management within an MPA might be to implement catch or gear restrictions, but we selected these categories because they were the most frequently cited in the literature. If multiple studies reported the same impact for a variable from the same MPA, only the most recent was used. Likewise, for no-take MPAs, if a study reported the same impact for the same variable at multiple points in time, only the most recent was included. However, if the impact of the variable differed between times for the same MPA, then both times were used. To assess the ecological recovery potential of periodic closures, pre-harvest, post-harvest, and recovery time points were all recorded separately. Finally, any additional factors suggested by authors in the studies as influencing MPA impacts were also recorded.

Impact evaluation techniques

To examine the extent to which the literature on impacts of South Pacific MPAs employs robust evaluation techniques, we examined each study's: (i) sampling design protocols, and (ii) justification of site selection and degree to which potential confounding factors were considered explicitly.

Most studies that intend to estimate MPA impact employ control-intervention (CI) or before-after-control-intervention (BACI) sampling protocols (Table 1), which might only

quantify outcomes, and not impact, if the underlying assumptions are not verified (Adams et al. 2019). A control-intervention approach assumes that there were no differences in the outcomes of interest between the control and intervention sites prior to the implementation of the intervention. The need for this assumption is avoided by a before-after-control-intervention (BACI) approach which assumes, however, parallel trends in variables of interest, that is, in the absence of the intervention the difference between the intervention and the control groups with respect to the outcome of interest is constant over time. An extension of the BACI approach is the paired series BACI (BACIPS), with which individual MPA sites are paired with control sites, which also assumes parallel trends in variables of interest. The degree to which these assumptions hold depends on how well potential confounders are accounted for. For example, matching methods (Table 1) provide the most rigorous approach to ensuring that confounders are accounted for, and can be applied to CI, BACI, and BACIPS sampling designs, although this approach has emerged only slowly in the MPA evaluation literature. Perception data from socioeconomic studies were also included in our review if the questions were framed so as to contain a reflexive counterfactual (Table 1), which involves framing survey questions in a way that attributes causality to an intervention. While reflexive counterfactuals can avoid the potential pitfalls of confounding factors, they also assume that each individual questioned has an accurate knowledge of the system both before and after intervention, as well as a strong understanding of attributing causality.

How well studies considered potential confounding factors was then assessed by searching publications for justification of selecting both MPA and control sites, as well as explicit recognition that additional factors could be masking actual impact. Many MPA evaluation studies often situate control sites immediately adjacent to MPAs without justification, assuming that this accounts for most potential confounders. We therefore included this approach as an additional category within our analysis. Studies were therefore categorized based on whether there was: (i) explicit discussion of site selection criteria and potential confounding factors; (ii) selection of spatial control sites immediately adjacent to MPA sites but with no mention of reasons; (iii) no discussion of site selection criteria or the potential for confounding factors to affect results; or (iv) clear evidence that the authors had selected biased control sites or the presence of additional confounding factors likely masking the true impact of MPAs (with or without discussion of site selection criteria). All data are available in the supplementary materials.

Results

Of the 87 articles that were selected for full text viewing, 52 studies examining the impacts of 65 MPAs satisfied the selection criteria and were analysed further (Fig. 2, Table 2). There was a large disparity in the number of cases assessing MPA impact between countries, with two countries—Fiji and New Caledonia—accounting for 75% of all studies. There was also a large disparity between the number of studies assessing ecological and socioeconomic variables. Of the 52 studies examined, only eight assessed socioeconomic data with methods that aimed to quantify impact.

Table 2 Key characteristics of the 52 studies evaluating MPA impact in the South Pacific

Citation	Country	Protected area name(s)	Data type	Governance type	Management type	MPA age (years)	MPA size (km²)	Experimental design	Site selection and confounding factors
Albert et al. (2016)	Fiji	Dravuni, Muaivuso and Namada	Ecological	CB	NT	2, 2, 3	Not listed, 3, 0.5	CI	Not discussed but adjacent control sites
Bartlett et al. (2009a)	Vanuatu	Nguna, Pele and Emao	Socioeconomic	CB	NT & PC	4 to 6	Not listed	CI	Explicit discussion of site selection
Bartlett et al. (2009b)	Vanuatu	Not listed	Ecological	CB	NT & PC	4 to 6	0.07 to 0.21	CI	Explicit discussion of site selection
Berdach (2003)	Tuvalu	Funafuti protected area	Ecological and Socioeconomic	C	NT	6	33	CI & RC	Not discussed
Beukering et al. (2014)	Fiji	Navakavu	Socioeconomic	CB	NT	5	37.1	CI	Explicit discussion of site selection
Bonaldo and Hay (2014)	Fiji	Namada, Vatu-o-lalai and Votua	Ecological	CB	NT	10, 10, 10	Not listed	CI	Explicit discussion of site selection
Bonaldo et al. (2017)	Fiji	Namada, Vatu-o-lalai and Votua	Ecological	CB	NT	10, 10, 10	Not listed	CI	Explicit discussion of site selection
Carassou (2013)	New Caledonia	Abore, Merlet and Bourail	Ecological	C	NT	8, 16 and 34	Not listed	CI with different fishing pressures	Not discussed
Clements and Hay (2017)	Fiji	Namada, Vatu-o-lalai and Votua	Ecological	CB	NT	Not listed	0.45 to 0.78	CI	Not discussed

Table 2 (continued)

Citation	Country	Protected area name(s)	Data type	Governance type	Management type	MPA age (years)	MPA size (km²)	Experimental design	Site selection and confounding factors
Clements and Hay (2018)	Fiji	Namada, Vatu-o-lalai and Votua	Ecological	CB	NT	12 to 13	0.45, 0.48, 0.78	CI	Explicit discussion of site selection
Clements et al. (2012)	Fiji	Komave, Namada,Namatakula, Votua	Ecological	CB	NT	4 to 6	0.5 to 0.8	CI	Not discussed but adjacent control sites
D'agata et al. (2016)	New Caledonia	Beautemps-Beaupre, Ouvea, Borendy, Pouebo, Hienghene, Southern Lagoons MPAs, Pines Island and Yves Merlet	Ecological	C & CB	NT & PC	Not listed, 21, 38	Not listed, 175	CI with gradient of controls	Explicit discussion of site selection
Dell et al. (2015)	Fiji	Namada, Vatu-o-lalai and Votua	Ecological	CB	NT	Not listed	0.5 to 0.8	CI	Explicit discussion of site selection
Dell et al. (2016)	Fiji	Vatu-o-lalai and Votua	Ecological	CB	NT	Not listed	0.5 to 0.8	CI	Explicit discussion of site selection
Dixson et al. (2014)	Fiji	Namada, Vatu-o-lalai and Votua	Ecological	CB	NT	Not listed	Not listed	CI	Explicit discussion of site selection
Dumas et al. (2010)	Vanuatu	Mangaronga and Marow	Ecological	CB	NT	3, 4	0,006 and 0.023	CI	Not discussed
Dumas et al. (2013)	New Caledonia	Southern Lagoon MPAs	Ecological	C	NT	19, 24, 32	Not listed	CI	Explicit discussion of site selection
Dumas et al. (2012)	Vanuatu	Anelcowat, Mangaliliu, Marou and Takara	Ecological	CB	PC	3, 7, 11, 12	0.033, 0.137, 0.15, 0.255	CI	Not discussed

Table 2 (continued)

Citation	Country	Protected area name(s)	Data type	Governance type	Management type	MPA age (years)	MPA size (km²)	Experimental design	Site selection and confounding factors
Egli et al. (2010)	Fiji	Kiobo, Nakorovou and Navatu	Socioeconomic	CB	NT	4	Not listed	BACI & CI	Not discussed
Ferraris et al. (2005)	New Caledonia	Abore	Ecological	C	NT	5	148	BACI for MPA removal	Explicit discussion of site selection
Goetze et al. (2011)	Fiji	Namena and Namuri	Ecological	CB	NT	4, 12	4.25, 60.6	CI	Evidence for biased control sites
Goetze et al. (2015)	Fiji	Kiobo and Natokalau	Ecological	CB	PC	5	2.07	BACI	Evidence for biased control sites
Goetze et al. (2017a)	Fiji	Nakodu, Natokalau, Nauouo, Tuatua	Ecological	CB	PC	3 and 7	0.73, 1.34, 2.17, 3.69	BACI	Not discussed but adjacent control sites
Goetze and Fullwood (2013)	Fiji	Namena	Ecological	CB	NT	12	60	CI	Evidence for biased control sites
Goetze et al. (2016)	Fiji	Kiobo, Nakodu, Natoka-lau, Nauouo, Tuatua	Ecological	CB	PC	3, 4, 7, 8	.73, 1.34, 2.07, 2.17, 3.69	BACI	Not discussed but adjacent control sites
Januchowski-Hartley et al. (2014)	Vanuatu	Not listed	Ecological	CB	NT & PC	1.5, 6	0.08 to 0.10	BACI	Not discussed but adjacent control sites
Jimenez et al. (2015)	New Caledonia	Not listed	Ecological	C	NT	10, 14, 19	Not listed	CI	Explicit discussion of site selection

Table 2 (continued)

Citation	Country	Protected area name(s)	Data type	Governance type	Management type	MPA age (years)	MPA size (km²)	Experimental design	Site selection and confounding factors
Jimenez (2016)	New Caledonia	Not listed	Ecological	C	NT	10, 14, 19	Not listed	CI	Explicit discussion of site selection
Jupiter and Egli (2011)	Fiji	Namena, Namuri, Nasue, Nakali, Yamotu Lase	Ecological	CB	NT & PC	2, 10	.13, .77, 4.24, 8.14, 60.6	CI	Not discussed but adjacent control sites
Jupiter (2010)	Fiji	Namena, Namuri, Nasue, Cakaulevu, Talai-i-lau, Vatuka, Nakali, Yamotu Lase	Ecological	CB	NT & PC	2, 3, 10	.13, .77, 4.24, 8.14, 14.05, 15.52, 18.85, 60.6	CI	Not discussed
Jupiter et al. (2012)	Fiji	Cakaulevu	Ecological	CB	PC	3	15.5	BACI	Evidence for biased control sites
Jupiter et al. (2013)	Fiji	Totoya, Moala, Tuvuca, Cicia, Vanuabalavu	Ecological	CB	NT	Not listed	Not listed	CI	Not discussed
Jupiter et al. (2017)	Fiji	Cakau Bavu, Cakau Naitaga, Cakaulevu, Nakali, Nakodu, Tuatua, Vatunibalagi and Yakuta	Ecological	CB	PC	3.5, 4, 5, 7, 8	0.2, 0.7, 1.3, 2.1, 3.7, 4.7, 15.5	CI	Not discussed but adjacent control sites
Kulbicki et al. (2007)	New Caledonia	Abore	Ecological	C	NT	5	148	BACI for MPA removal	Explicit discussion of site selection
Lalavanua et al. (2014)	Fiji	Batiki	Ecological	CB	NT	Not listed	0.02	CI	Not discussed
Langlois et al. (2006)	New Caledonia	Southern Lagoon MPAs	Ecological	C	NT	Not listed	Not listed	CI	Not discussed but adjacent control sites

Table 2 (continued)

Citation	Country	Protected area name(s)	Data type	Governance type	Management type	MPA age (years)	MPA size (km²)	Experimental design	Site selection and confounding factors
Léopold et al. (2009)	Fiji	Navakavu	Ecological	CB	NT	3	Not listed	CI	Explicit discussion of site selection
Moore et al. (2013)	Tuvalu	Funafuti protected area	Ecological	C	NT	17	33	CI	Not discussed
Pascal (2009)	Vanuatu	Emura, Piliura, Unakap, Laonamoa and Worasifiu	Socioeconomic	CB	NT	4 to 6	0.12 to 0.24	CI	Explicit discussion of site selection
Pascal and Seidle (2013)	Fiji and Vanuatu	Emua, Siviri, Tanoliu, Mangaliliu, Tassiriki, Namada, Tagaqe, Vatu-o-lailai, Votua and Navakavu	Socioeconomic	CB	PC	Not listed	Not listed	CI	Unclear
Peters (2017)	Fiji	Bikini and Rabbit	Ecological	CB	NT	Not listed	Not listed	CI	Not discussed
Powell et al. (2016)	New Caledonia	Southern Lagoon MPAs	Ecological	C	NT	18	Not listed	CI	Not discussed
Preuss et al. (2009)	New Caledonia	Abore	Ecological	C	NT	3 to 11	50	BACI for MPA removal	Explicit discussion of site selection
Rasher and Hay (2010)	Fiji	Votua	Ecological	CB	NT	4	Not listed	CI	Not discussed
Rasher (2013)	Fiji	Namada, Vatu-o-lalai and Votua	Ecological	CB	NT	11	Not listed	CI	Not discussed
Siaosi et al. (2012)	Tuvalu	Funafuti protected area	Ecological	C	NT	15	33	CI	Not discussed
Thaman et al. (2017)	Fiji	Navukavu	Socioeconomic	CB	NT	Not listed	Not listed	RC	Unclear

Table 2 (continued)

Citation	Country	Protected area name(s)	Data type	Governance type	Management type	MPA age (years)	MPA size (km²)	Experimental design	Site selection and confounding factors
Thiault et al. (2019)	French Polynesia	Moorea MPA network	Ecological	C	NT	6	Not listed	Progressive change BACIPS	Explicit discussion of site selection
Tran et al. (2016)	French Polynesia	Pihaena	Ecological	C	NT	10	0.578	CI	Not discussed but adjacent control sites
Wantiez et al. (1997)	New Caledonia	Southern Lagoon MPAs	Ecological	C	NT	5	Not listed	BACI	Evidence for biased control sites
Webster et al. (2017)	Tonga	O'ua	Socioeconomic	CB	NT	Not listed	Not listed	RC	N/A
Zimmerman et al. (2018)	Fiji	Rabbit	Ecological	CB	NT	Not listed	Not listed	CI	Not discussed

Governance type CB refers to community-based and C to central; management strategy NT refers to no-take and PC to periodic closures. See Table 1 for definitions

Impacts and related factors

Impacts

Six hundred and sixty-two instances of 151 ecological impact variables were recorded. Overall, 42% of instances reported positive ecological impacts. The most frequently measured variables were total fish diversity and target fish biomass, and the nine most frequently measured variables accounted for 41% of the measured impacts (Fig. 3). While 50% of studies that reported total fish biomass indicated positive impacts, only 38% were positive in the case of target fish biomass. The inverse was true for fish density (total 41%; target 48%). Positive impacts for invertebrates (67%) were almost twice as numerous as those measured for fish (38%). Positive impacts on coral cover were recorded in only 13% of cases.

Seventy-six instances of 49 socioeconomic variables were recorded. Overall, 72% of these reported positive socioeconomic impacts (Fig. 3). Socioeconomic variables were grouped into five categories for summary analysis: (i) catch (e.g. CPUE, maximum catch size); (ii) economic impacts (e.g. income growth, revenue from tourism); (iii) resource management decision-making (e.g. participation, inclusion of marginalised groups); (iv) perceptions of ecological change (e.g. perception of coral cover, fish biomass); and (v) perceptions of socioeconomic change (e.g. perceived change in remittance, change to income from fishing). All five categories had generally positive impacts, most frequently for catch, economic impacts, and perceived socioeconomic benefits. Neutral perceptions
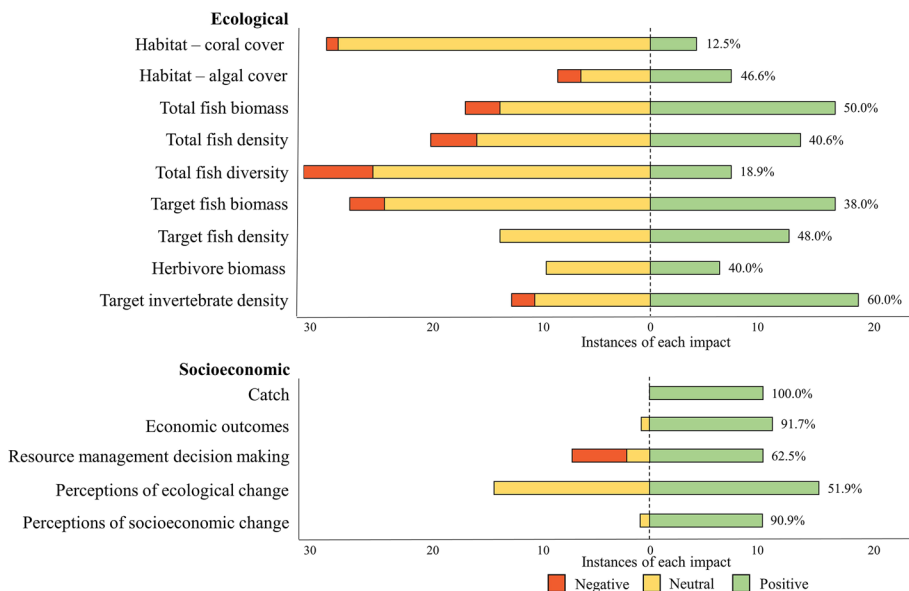

Fig. 3 Positive, neutral, and negative impacts of MPAs. Ecological impacts shown include only a subset of the most frequently measured 151 ecological variables. Decreasing algal cover was considered a positive ecological impact. Socioeconomic impacts containing 76 study variables were divided into five groups. Numbers to the right of each bar indicate the percentages of measured impacts that were positive. Neutral impacts were included to the left of zero because MPAs aim to create positive impact

of ecological change were reported most frequently for changes in fish abundance, size and diversity, habitat health, and giant clam abundance. The most frequent negative impacts were recorded for participation which, compared to control villages, comprised four studies in which community members reported less ability to participate in meetings or have their interests represented.

Factors related to MPA success and failure

Both centrally governed and community-based MPAs had similar percentages of positive ecological impacts (48% and 43% respectively) (Fig. 4), while socioeconomic impact variables were largely positive, regardless of the governance approach or management strategy (Fig. 4). Community-based governance was the most commonly measured governance type. Thirty-six studies, examining 43 MPAs, measured the impact of community-based governance (both no-take and periodic closures), compared to 15 studies examining 14 MPAs that assessed impacts from central governance. These numbers were biased by country, with most centrally governed studies originating in New Caledonia and most community-based studies coming from Fiji and Vanuatu.

The greatest percentage of neutral and negative ecological impacts was for periodic closures (71%), which were implemented only under community-based governance approaches. Five studies examining 10 MPAs quantified the impacts of harvesting and recovery on periodic closures (Fig. 5). Pre-harvest and post-harvest measurements were typically taken within 1 month each side of harvesting events, while recovery was measured 1 year later. There was a clear decline in the number of positive ecological impacts after harvest events and limited instances of recovery. In only two instances did a variable
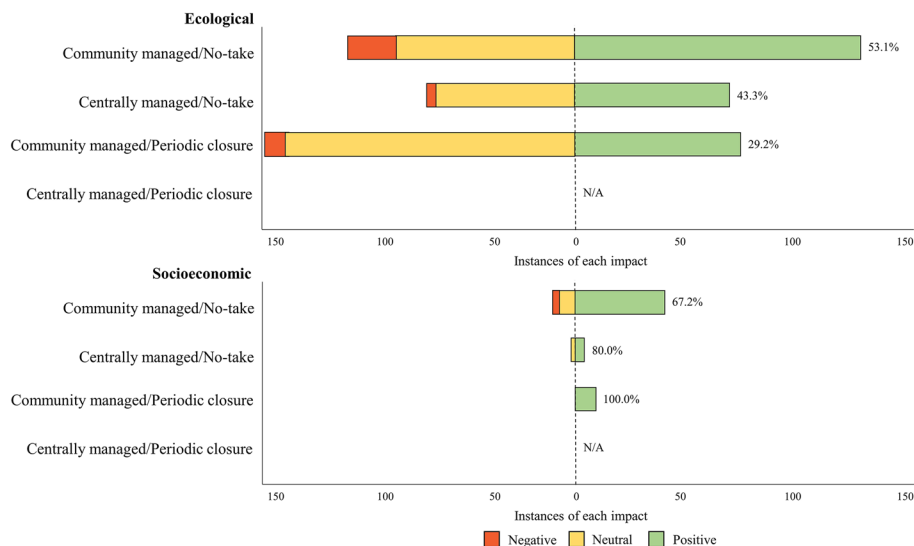

Fig. 4 Positive, neutral, and negative ecological and socio-economic impacts in relation to governance and management strategies of MPAs. Numbers to the right of each bar indicate the percentages of measured impacts that were positive. Neutral impacts were included to the left of zero because MPAs aim to create positive impact

have a positive ecological impact following recovery and these were for the biomass of low and moderately vulnerable fish species.

Centrally governed MPAs were larger (mean 81 km^2±29.3 SE) and older (mean 12 years±9.84 SE) than community governed MPAs (mean size 5 km^2±1.9 SE; mean age 5 years±0.4 SE). There was no significant correlation between the proportion of positive impacts and either the age or size of MPAs (age: r=0.238, n=47, p=0.107; size: r=−0.030, n=45, p=0.844). However, the mean percentage of positive impacts for MPAs less than ten years old was 36% (±4.9 SE), while for MPAs greater than 10 years old it was 67% (±6.0 SE).

Few studies discussed additional factors associated with positive MPA impacts beyond those listed above (governance, management, age, and size). However, 26 additional factors suggested to explain neutral or negative MPA impacts were identified in studies that did not observe expected positive impacts (Fig. 6). These additional factors can be divided broadly into six categories, the relative frequency of which varied between governance approach and management strategy. When centrally governed MPAs failed to achieve positive impacts, it was generally suggested that the reasons were environmental (e.g. sediment discharge from a river mouth) or biological (e.g. changing predator dynamics). In contrast, when community-based MPAs failed to achieve positive impacts, factors most often suggested were related to reserve design (e.g. close to human populations), management (e.g. lack of compliance), or social constraints (e.g. poacher aggression). Compared with no-take reserves, failure of periodic closures to achieve positive impacts was suggested to be more likely associated with reserve design.

Impact evaluation techniques

Most studies (73%) used control-intervention techniques (Table 2, Fig. 7). No studies used only before-after data. Of the 52 studies, only 21 explicitly discussed any potential confounding factors in the selection of MPA and control sites. Within the studies that provided explanations for site selection, the reasoning was exclusively ecological; no studies considered any potential socioeconomic confounding variables in their sampling design. Of those that discussed ecological variables, the predominant consideration was habitat
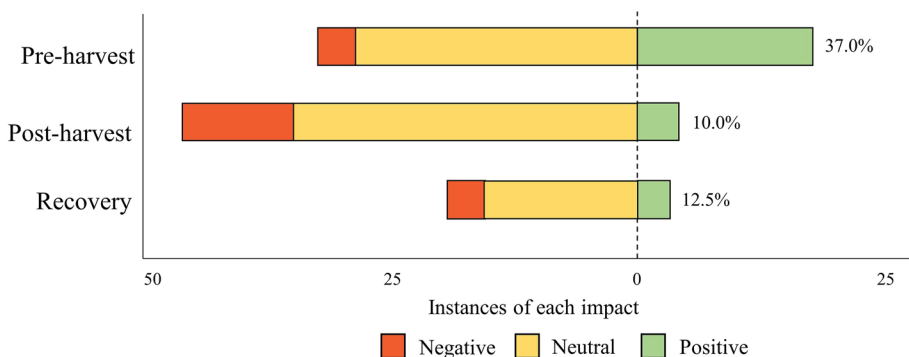
Fig. 5 Numbers of ecological variables measured with positive, neutral, or negative impacts for periodic closures. Results are shown for pre-harvest (<1 month), post-harvest (<1 month), and following a recovery period (~1 year). Numbers to the right of each bar indicate the percentages of measured impacts that were positive. Neutral impacts were included to the left of zero because MPAs aim to create positive impact

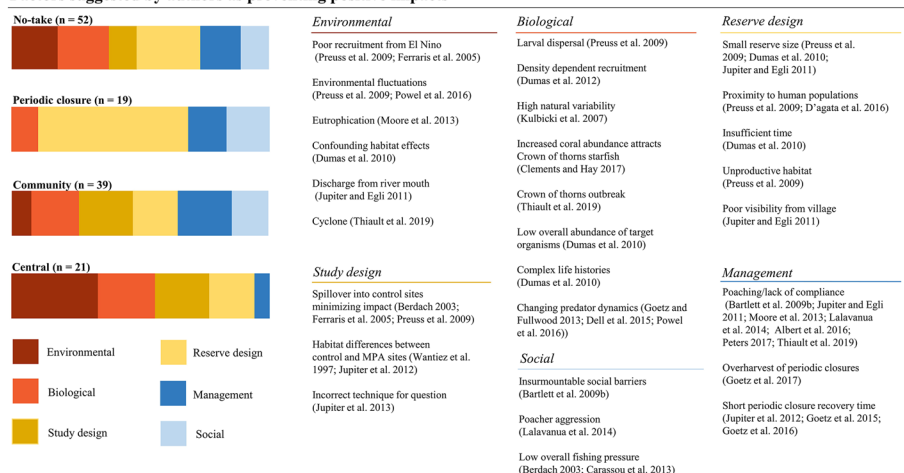**Factors suggested by authors as preventing positive impacts**

No-take (n = 52)

Periodic closure (n = 19)

Community (n = 39)

Central (n = 21)

Environmental

Biological

Study design

Reserve design

Management

Social

Environmental

Poor recruitment from El Nino (Preuss et al. 2009; Ferraris et al. 2005)

Environmental fluctuations (Preuss et al. 2009; Powel et al. 2016)

Eutrophication (Moore et al. 2013)

Confounding habitat effects (Dumas et al. 2010)

Discharge from river mouth (Jupiter and Egli 2011)

Cyclone (Thiault et al. 2019)

Study design

Spillover into control sites minimizing impact (Berdach 2003; Ferraris et al. 2005; Preuss et al. 2009)

Habitat differences between control and MPA sites (Wantiez et al. 1997; Jupiter et al. 2012)

Incorrect technique for question (Jupiter et al. 2013)

Biological

Larval dispersal (Preuss et al. 2009)

Density dependent recruitment (Dumas et al. 2012)

High natural variability (Kulbicki et al. 2007)

Increased coral abundance attracts Crown of thorns starfish (Clements and Hay 2017)

Crown of thorns outbreak (Thiault et al. 2019)

Low overall abundance of target organisms (Dumas et al. 2010)

Complex life histories (Dumas et al. 2010)

Changing predator dynamics (Goetz and Fullwood 2013; Dell et al. 2015; Powel et al. 2016))

Social

Insurmountable social barriers (Bartlett et al. 2009b)

Poacher aggression (Lalavanua et al. 2014)

Low overall fishing pressure (Berdach 2003; Carassou et al. 2013)

Reserve design

Small reserve size (Preuss et al. 2009; Dumas et al. 2010; Jupiter and Egli 2011)

Proximity to human populations (Preuss et al. 2009; D'agata et al. 2016)

Insufficient time (Dumas et al. 2010)

Unproductive habitat (Preuss et al. 2009)

Poor visibility from village (Jupiter and Egli 2011)

Management

Poaching/lack of compliance (Bartlett et al. 2009b; Jupiter and Egli 2011; Moore et al. 2013; Lalavanua et al. 2014; Albert et al. 2016; Peters 2017; Thiault et al. 2019)

Overharvest of periodic closures (Goetz et al. 2017)

Short periodic closure recovery time (Jupiter et al. 2012; Goetz et al. 2015; Goetz et al. 2016)

Fig. 6 Additional factors suggested by authors of reviewed studies when MPAs failed to achieve positive impact. Factors, grouped into six categories, are allocated according to management (no-take MPAs or periodic closure) and governance (community-based or central). The sample size (n) indicates the number of studies included in each category

and, in a few cases, wave energy. While 20 studies selected control sites immediately adjacent to the MPAs, nine of these did so without explicit statements about what factors were being controlled for. Fourteen of the studies did not discuss the selection of control sites at all. Lastly, in five studies, it was clear that confounding factors were present that could
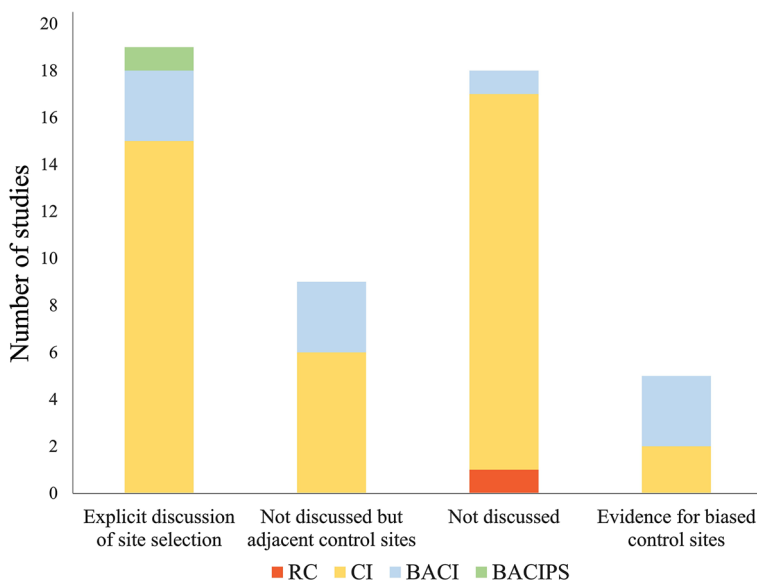
Number of studies

Explicit discussion of site selection

Not discussed but adjacent control sites

Not discussed

Evidence for biased control sites

RC   CI   BACI   BACIPS

Fig. 7 Evaluation of study quality, based on experimental design and criteria for site selection and consideration of confounding factors. See Table 1 for definitions

influence outcome variables, potentially causing over- or under-estimations of true impact (Table 2, Fig. 7).

Discussion

Study designs, methods for estimating impact of MPAs, and uncertainties around impact estimates can be understood with a theory of change that illustrates their relationships (Fig. 8). In the sections that follow, we explore the implications of our findings and how they relate to different aspects of this theory of change. Specifically, we discuss: (i) direct and indirect ecological and socioeconomic impacts; (ii) factors related to MPA success and failure; and (iii) the extent to which counterfactual thinking has been embraced by impact evaluation programs in the region.

Overall, we found that only half of all measured impacts of MPAs in the South Pacific were positive and, although from far fewer studies, the proportion of positive impacts was greater from socioeconomic studies. Community-based and centrally governed MPAs also had similar proportions of positive impacts, suggesting that both governance approaches are viable options in the region. Positive impacts were more common for no-take MPAs than periodic closures, and there was limited evidence of any ecological recovery potential in periodic closures following harvesting events. Although most of the reviewed MPAs had not been implemented for long, those that were older than 10 years had a higher proportion of positive impacts. A wide range of factors were reported by authors as being related to neutral or negative impacts, and these differed between management strategies and governance approaches. However, all the results of this study must be considered in the context of the MPA evaluation literature from the region rarely embracing explicit counterfactual thinking.

Impacts and related factors

Direct and indirect ecological impacts

The most commonly measured ecological impacts were fish biodiversity and target species biomass, likely reflecting broad conservation and local community objectives respectively (Jupiter et al. 2014). However, the neutral and negative results for these two variables in the majority of studies indicated that MPAs in the region failed to achieve these objectives more than 50% of the time. MPAs are a management strategy that directly affects only target species (Mosquera et al. 2000), so the impacts of MPA implementation should be most evident in these organisms. All other outcomes will depend largely on changes in ecosystem dynamics based on the response of target species (Allison et al. 1998). It is therefore important to understand why, in many instances, MPAs failed to increase target species populations. Of the 47 MPAs in our study with a known age, 76% were less than 10 years old, which, considering the long recovery times for many target species, could account for these poor results.

Benthic cover is rarely affected directly by MPA implementation, except where extensive damage occurs from anchoring, destructive fishing practices, collecting, or development (Milazzo et al. 2004). Rather, indirect mechanisms (Fig. 8) by which MPAs can affect benthic cover are primarily through increases in herbivory, which reduces the competitive dominance of algal assemblages on corals (Lirman 2001; McCook et al. 2001; Hughes
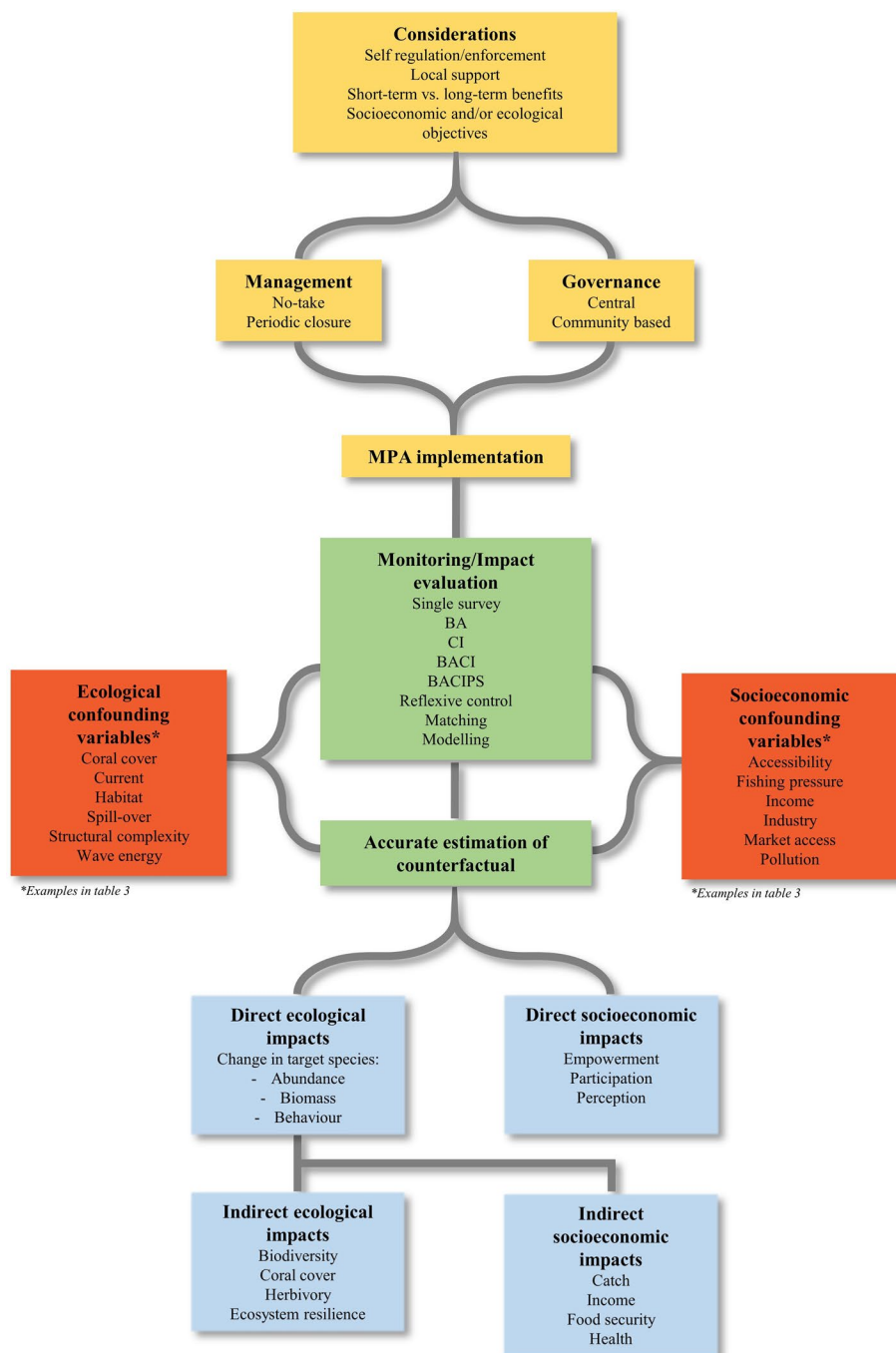
et al. 2007). However, while the relationship between coral-algal interactions and herbivory
is well documented, few studies have demonstrated the ability of MPAs to change these
relationships (but see Rasher and Hay 2010; Bonaldo and Hay 2014; Dell et al. 2016). This
lack of evidence might be driven by discrepancies between short funding cycles for moni-
toring MPA impact and the time required for changes in coral cover to occur. Ultimately, as
herbivores increase after protection, the balance between algal and coral dominance should
shift. However, these results might take decades to manifest (Abesamis et al. 2014) and
might also be masked by additional confounding factors that affect coral-algal interactions,
such as wave energy (Adey 1998) or nutrient levels (McManus and Polsenberg 2004).

The greatest percentage of positive ecological impacts was found for the density of
target invertebrates. In the South Pacific, invertebrates are often highly targeted and eas-
ily harvested, making them vulnerable to overexploitation (Uthicke and Conand 2005).
However, given that many species reproduce and mature quickly (Battaglene 1999) and
have small home ranges (Purcell and Kirby 2006), they also have a high potential for rapid
recovery following harvest reductions. Nonetheless, these life-history characteristics have
also resulted in the massive overharvest and functional collapse of several target inver-
tebrate populations in the South Pacific (Conand et al. 2003). For MPAs to be effective
at allowing stock recovery of target invertebrates, it is critical that meta-populations are
sufficiently intact to allow recruitment into the protected areas after closure (Uthicke and
Conand 2005).

Direct and indirect socioeconomic impacts

Most of the recorded socioeconomic impacts identified were positive, which suggests that,
for the South Pacific, MPAs can be a viable strategy for both conservation and develop-
ment. Given that some socioeconomic impacts are not mediated by ecosystem change
(Fig. 8; Gurney et al. 2014), they can likely manifest over much shorter periods of time,
which, given the young age of most MPAs, could explain the higher percentage of positive
socioeconomic than ecological impacts. In addition, perceptions of ecological change, a
socioeconomic impact, might not always be aligned with actual changes in the environ-
ment. For example, Bartlett et al. (2009a) and Yasue et al. (2010) highlighted how per-
ceptions of ecological variables are generally much greater than quantified ecological
outcomes. Despite these considerations, the results suggest that the evidence for positive
social impacts from MPAs in the South Pacific is strong.

Our review identified only eight studies that quantified socioeconomic impacts. Many
socioeconomic studies in the South Pacific have focused on factors leading to successful
MPA implementation (e.g. Govan et al. 2006; Abernethy et al. 2014; Cohen et al. 2014) or
discussed the importance and revitalization of traditional management (e.g. Johannes 1978,
2002; Govan 2009a). The relatively small number of impact studies likely arises from key
challenges that make quasi-experimental designs difficult to implement in social science
research. These challenges include achieving a sufficient sample size, particularly at the
level of villages, and finding appropriate control villages, which are both similar to MPA

villages and have people willing to be used as controls. Outcomes can also vary between subgroups (Gurney et al. 2015), and inequality among social groups can lead to conflict (Fabinyi et al. 2013), jeopardizing achievement of goals for both social and ecological projects (Persha and Andersson 2014). Because of these problems, traditional control-intervention and BACI sampling designs are somewhat less feasible with social data (but see included studies and Gurney et al. 2014, 2015). An additional caveat on the results of this review is that favouring quasi-experimental designs and quantitative data might also risk ignoring many potential impacts of MPAs on people that are not easily quantified. Such impacts are likely to be related to non-material connections between humans and nature (such as cultural and relational values; e.g. Chan et al. 2016; Lau et al. 2019) that are increasingly emphasised in recent literature. An example of a conceptual framework that includes non-material values is that of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, emphasising nature's contribution to people (Díaz et al. 2018). Therefore, assessments of the socioeconomic impacts of MPAs should ideally take a mixed-methods approach (e.g. Sterling et al. 2017), drawing on both quantitative and qualitative methods, while recognising the inherent limitations of both.

Factors related to MPA success and failure

Of the two governance approaches examined (Fig. 8), community-based governance, which can be established for a range of purposes (e.g. food security, maintaining traditional tenure), had similar proportions of positive impacts as centrally governed MPAs, which are ostensibly focused on nature conservation (Figs. 4, 6). While community-based MPAs are rarely systematically configured to maximize impact, their configurations can still be close to optimal (Smallhorn-West et al. 2018). This is because community-based MPAs are often situated close to villages for social reasons, such as ease of enforcement, which can result in higher impacts in otherwise heavily fished areas. In contrast, while centrally governed MPAs have the potential to be systematically configured to achieve the greatest impact, in practice they can often be situated residually (Devillers et al. 2015) where impacts can be limited.

Of the two management strategies examined (Fig. 8), no-take MPAs, which were generally more effective than periodic closures (Fig. 4), often have straightforward enforcement and simpler regulations, with clear benefits accruing both inside no-take reserves and from spillover to adjacent areas (Abesamis and Russ 2005). Given the direct objectives of MPAs are typically to increase target species biomass and density, it is also clear that strategies that minimize harvest effort should have the greatest conservation impacts. However, while there is a general consensus on the greater potential benefits of no-take MPAs, Giakoumi et al. (2017) suggested that periodic closures can still be useful for heavily human-dominated regions because multiple users have interests that are often in conflict, and no-take MPAs can be considered obstacles to some of their activities. This is further supported by the results of Bartlett et al. (2009b), who found that periodic closures were more effective ecologically than no-take MPAs. These authors suggested that no-take MPAs in the Asia–Pacific region commonly fail to meet their objectives due to low compliance (McClanahan et al. 2006) and insurmountable social barriers (Cinner and Aswani 2007). They concluded that, in the community context, periodic closures can provide an acceptable alternative to no-take reserves because they are both practical and locally appropriate.

Our review also suggests that the ecological benefits of periodic closures are limited to pre-harvest conditions, when they are effectively acting as recently implemented no-take

MPAs, with little evidence of any post-harvest recovery. The recovery time following highly intensive harvesting events can be between 5 and 20 years (Abesamis et al. 2014), much greater than the 1-year post-harvest timing used by most studies to measure recovery (Jupiter et al. 2012; Goetze et al. 2016). Periodic closures are therefore most likely to achieve short-term objectives such as increasing fisheries yields from single, repeated harvest events, and are unlikely to achieve longer-term conservation objectives (Goetze et al. 2017b).

The equal proportion of positive impacts for MPAs regardless of size indicates that small MPAs can be effective and that size should not be the sole consideration during the design phase. Residually situated reserves with low impact can be larger in size and more typical of centrally governed areas, because proclaiming MPAs in areas of little value to resource extraction industries is likely to face little opposition whilst providing a means for governments to apparently fulfil their conservation commitments. In contrast, smaller, community governed reserves can be configured in less residual areas where potential impact can be higher (Smallhorn-West et al. 2018). Further studies should clarify the trade-offs between reserve size and management strategies while accounting for differences in potential impact.

The factors suggested to account for MPAs failing to achieve positive impacts differed between management and governance strategies. While this disparity could result from differences in the characteristics of the reserves themselves, the articles reviewed did not suggest noticeable ecological or socioeconomic differences in MPAs between governance or management types. An alternative explanation is that, when MPAs fail to have a positive impact, researchers studying community governed MPAs might focus more on socioeconomic factors while those studying centrally governed MPAs might give more consideration to biological or environmental factors.

Impact evaluation techniques

While acknowledging that many MPA studies are opportunistic, it is clear that, in the South Pacific, counterfactual thinking has yet to be fully embraced and that more consideration is needed of the potential for confounding factors to obscure actual impacts (Fig. 8). Therefore the results of part 1 of this review must be considered while acknowledging the limited efficacy of most studies to quantify actual impacts. The non-random placement of MPAs can result in biases towards specific locations (e.g. high-quality environments, residual areas), leading to over- or under-estimations of impact. While control sites selected by many of the included studies could, in reality, represent fairly accurate counterfactual conditions, unless these conditions are quantified explicitly, or at the very least clearly considered, it is difficult to attribute causality to MPA implementation. No studies in the South Pacific quantitatively accounted for confounding factors, and 60% of studies did not explicitly discuss any selection criteria for MPA or control sites. There was some evidence of a trend for studies to select control sites with a similar habitats as MPA sites, but few other ecological factors were expressly considered. Furthermore, there is no evidence that any potential socioeconomic confounders were considered during the selection of survey sites. This result is particularly relevant given the growing body of literature demonstrating the key role of social dynamics in MPA impacts (e.g. Pollnac et al. 2010). Lastly, there may also be more general biases towards better performing MPAs being published in the literature over those with neutral or negative impacts.

Thiault et al. (2019), utilizing the BACIPS approach, provided the most robust methods used to date in the South Pacific and did so while explicitly discussing estimates of counterfactual conditions. The conceptual framework for this study is therefore an ideal starting point for researchers aiming to develop sound impact evaluation programs in the South Pacific. However, even within this well planned study, the caveat remains that pairing appears to be based exclusively on geographic proximity and physical characteristics, with no mention of socioeconomic conditions, and it is unclear whether pairing was quantified or subjective.

Table 3 provides examples of how confounding factors might lead to over- or under-estimations of impact in this region. Five studies in this review had clear potential for confounding factors to mask the true impact of MPAs. Wantiez et al. (1997) compared five control sites situated adjacent to the capital of New Caledonia with five MPA sites located several to tens of kilometres away. Observed differences between sites could be due to population pressure or fishing pressure and not the MPAs per se, resulting an overestimation of impact. Jupiter et al. (2012) discussed the potential confounding factor of large habitat differences between two of their control sites and two of their four MPA sites, although they noted that the location and replication of survey sites was constrained by the opportunistic nature of the study. Goetze et al. (2011), and subsequently Goetze and Fullwood (2013) and Goetze et al. (2015), specifically selected control sites so that "fished [control] sites were placed in areas adjacent to the reserves where high levels of fishing are known to occur". Control sites with high fishing pressure represent an accurate counterfactual only if the MPA sites also would have had equally high fishing pressure in the absence of management. This sampling design could over-estimate impact by failing to account for potential differences in fishing pressure in the absence of MPA implementation, such as low extractive potential of the MPAs. More generally, our review also found poor study design listed as a factor influencing observed neutral and negative MPA impacts, which indicates both that substantial improvements could be made to standard protocols and that the authors were aware of their studies' limitations. These results highlight the lack of systematic process in the current MPA impact evaluation literature by which accurate estimates of counterfactuals are produced.

In addition to these five studies, nine studies from Fiji compared various impacts between three MPAs and adjacent control sites in Namada, Vatu-o-lailai, and Votua. These MPAs are exceptional in having some of the greatest percentages of positive impacts recorded, particularly on ecosystem processes (rates of herbivory, crown-of-thorns starfish abundance) and benthic cover (210–280% greater coral cover inside MPAs). However, given their small size (~0.5 km^2) and that were only recently implemented at the time of data collection (many < 10 years), it would be necessary to demonstrate conclusively that the results were due to MPA implementation and not influenced by confounding factors such as a bias in reserve placement over high-quality habitats. While Bonaldo and Hay (2014) mentioned unpublished data reporting low coral cover in both MPA and control sites prior to implementation, the exceptional degree of impact reported underlines the need for all potential confounders to be considered.

Ways forward

With the current trend of rapid ecosystem degradation, researchers must often be both opportunistic when developing research methods and quick to draw robust conclusions from management interventions. Nonetheless, environmental policy must be

Table 3 Examples of potential ecological and socioeconomic confounding factors that can influence estimates of the difference between MPA and counterfactual conditions

Potential confounders	Examples of how poorly chosen control sites can lead to over- or under-estimation of impact
Coral cover and structural complexity	Greater coral cover and complexity increases the carrying capacity of an ecosystem. An MPA is configured to protect areas with exceptional coral cover. Subsequent control-intervention studies that fail to account for high coral cover will overestimate impact
Displaced fishing effort	An MPA displaces current fishing activity to a nearby reef, which is subsequently used as a control site. Displaced fishing effort from the MPA will result in variables of interest declining in nearby areas, with overestimation of impact, even though the net stock remains the same
Education	Education about ecological recovery is introduced by an NGO along with an MPA. Perceptions of ecosystem health in the MPA community therefore increase. At the same time they also conduct educational outreach in a nearby control village with no MPA, thereby increasing their understanding of the damage fishing is causing. Impact is overestimated because the difference in perceived change between MPA and control villages is the result of additional educational programs and not the implementation of the MPA
Fishing pressure	Control sites are selected in areas with higher fishing pressure than would have occurred in MPAs, overestimating impact. Sites with high fishing pressure do not represent an accurate counterfactual unless the MPA sites would also have had equally high fishing pressure in the absence of management. (e.g. Wantiez et al. 1997; Goetze et al. 2011, 2015; Goetze and Fullwood 2013)
Habitat quality	High/Low-quality habitats are selected for protection by MPAs, which have a higher/lower carrying capacity of target species than control sites. Subsequent control-intervention studies over/under-estimate impact. (e.g. Jupiter et al. 2012)
Income	A village with high average income is used as a control for an MPA village with low income. Fishing in the high-income village is conducted with new equipment and faster boats than the MPA village. Economic impact is underestimated because of failure to account for difference in fishing efficiency
Industry	A tuna canning factory is introduced near a village heavily reliant on fishing. The factory employs people from a nearby village with an MPA but not from the village acting as the control. Dependence on fishing decreases in the MPA village but remains stable in the control village. Income rises in the MPA village. The biological impact of the MPA is overestimated because the number of people fishing in the MPA village has decreased. The economic impact of the MPA is overestimated because increased income stems from employment in the factory
Market access	A non-MPA village has excellent access to a large market in the capital city. A nearby MPA village has greater catch rates, but economic impact is underestimated because they receive less income for their catch due to unequal market connection
Politics	A recent election has empowered many community members in an MPA village to participate in village affairs. Social impact of the MPA is overestimated because empowerment was not the result of the MPA, but of the recent election

Table 3 (continued)

Potential confounders	Examples of how poorly chosen control sites can lead to over- or under-estimation of impact
Pollution	Sedimentation from a nearby agricultural enterprise has increased algal proliferation on an MPA reef. Impact is underestimated compared to a healthy control site
Spillover from adjacent MPA	Control sites are located too close to MPA, within the radius of target species spillover. Surveys record a smaller difference between control and MPA sites and ultimately underestimate impact
Wave energy and current	High-current environments (e.g. lagoon entrances) can have greater abundances of fish than surrounding areas. An MPA is in the middle of a reef but the lagoon entrance is used as a control site. Greater species abundance at the lagoon entrance results in an underestimation of impact

evidence-based, and it is therefore imperative that either rigorous protocols are in place to demonstrate impact, or the implications of alternative evaluation methods are understood (Pressey et al. 2015; McIntosh et al. 2017; Pressey et al. 2017). Care must be taken to effectively manage two types of confounders, those that influence the observed variables themselves (e.g. effects on coral cover or fish biomass) and those that influence the placement of MPAs (e.g. residual locations with low inherent fishing pressure, or proximity to communities, where fishing pressure is high, for ease of enforcement). Ferraro (2009) and Ferraro and Hanauer (2014) provided the foundation for counterfactual thinking and impact evaluation in evaluating protected areas, and proposed both experimental and quasi-experimental designs to build the evidence base for the impact of environmental policy and conservation interventions. One such approach is the statistical matching of MPA sites to controls which, for ecological impacts of MPAs, is described in detail in Ahmadia et al. (2015). This approach can be used after MPA establishment to avoid observable selection bias and identify comparable control sites to accurately estimate counterfactual conditions (see R matching packages Matching and Matchit). Matching can therefore be used in both BACI (including BACIPS) and, when temporal data are not available, CI programs. A matched BACIPS approach (Thiault et al. 2019) would represent the most robust non-experimental method to determine MPA impact.

However, we acknowledge that employing matching methods require specialist statistical and coding expertise and that training in such might be difficult to access by researchers and practitioners based in the South Pacific. We suggest that further research should focus on developing simpler techniques for the preliminary matching of MPA and control sites based on predefined variables, or ways to easily tabulate the most important ecological and socioeconomic factors that could influence the variables being measured. An important starting point, not requiring specialist expertise, is to carefully consider both the potential ecological and socioeconomic factors that influence the variables of interest and placement of MPAs during site selection (Fig. 8), and to discuss these explicitly in subsequent publications. This approach would increase the robustness and clarity of conclusions regarding impact. To this end, we argue that MPA evaluation programs in the South Pacific should move towards fully embracing counterfactual thinking to allow researchers, managers, and stakeholders to draw robust conclusions regarding the difference made from both current and future marine protected areas.

Compliance with ethical standards

Conflicts of interest The authors declare there are no conflicts of interest.

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Affiliations

Patrick F. Smallhorn-West[1,2] · **Rebecca Weeks**[1] · **Georgina Gurney**[1] · **Robert L. Pressey**[1]

✉ Patrick F. Smallhorn-West
  patrick.smallhornwest@my.jcu.edu.au

[1] Australian Research Council Centre of Excellence for Coral Reef Studies, James Cook University, Townsville, QLD 4811, Australia

[2] Marine Biology and Aquaculture, College of Science and Engineering, James Cook University, 1 James Cook Drive, Townsville, QLD 4811, Australia