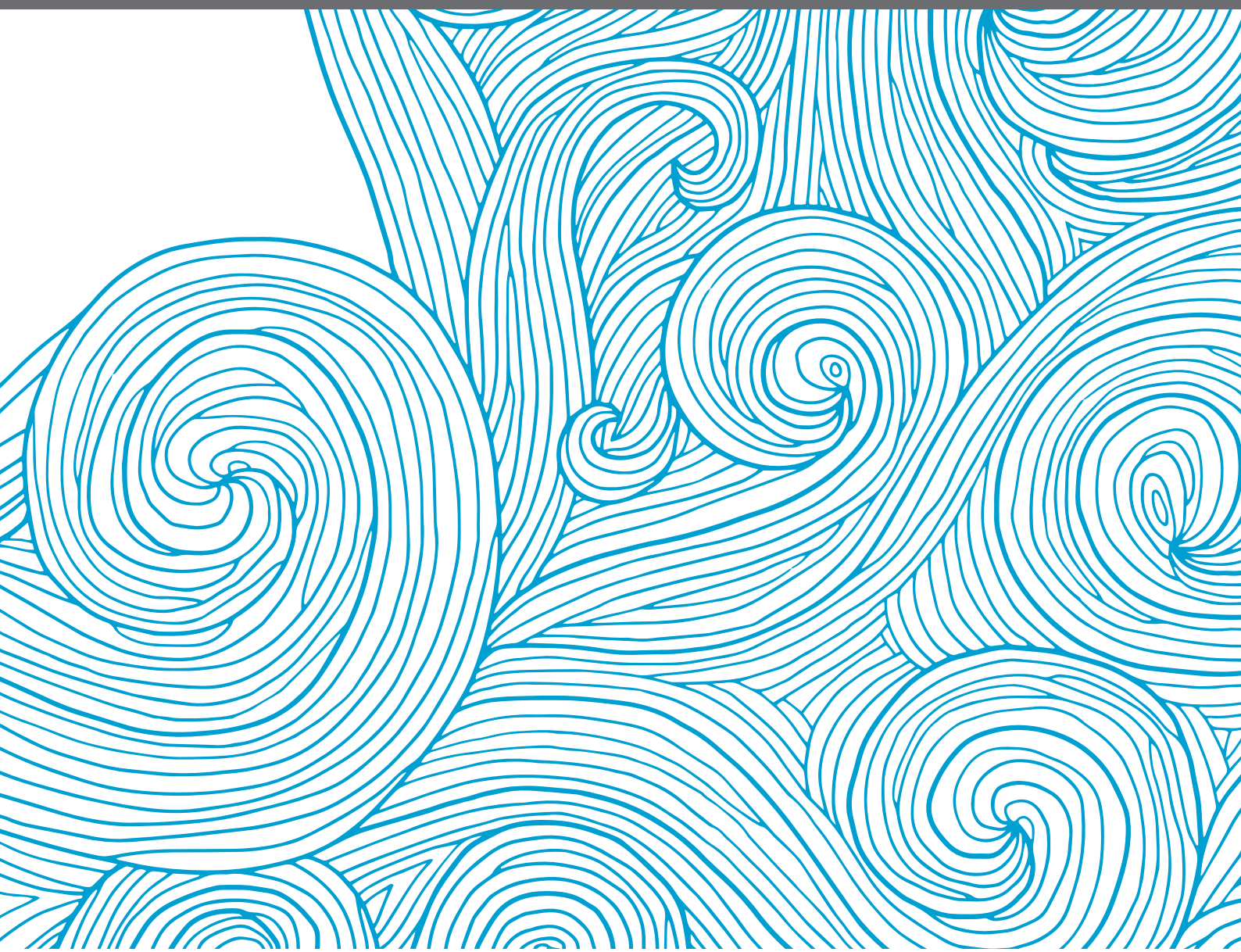
5TH INTERNATIONAL MARINE CONSERVATION CONGRESS

EDITED BY: E. Christien Michael Parsons, Edward Jeremy Hind-Ozan, Leslie Cornick, Mariana Padron and Jason Paul Landrum

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Topic Editors:
E. Christien Michael Parsons, National Science Foundation (NSF), United States
Edward Jeremy Hind-Ozan, Department for Environment, Food and Rural Affairs, United Kingdom
Leslie Cornick, University of Washington Bothell, United States
Mariana Padron, UMR8222 Laboratoire d'Ecogéochimie des Environnements Benthiques (LECOB), France
Jason Paul Landrum, Lenfest Ocean Program, Pew Charitable Trusts, United States

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OPINION
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Lions, Whales, and the Web: Transforming Moment Inertia into Conservation Action

Andrew D. Thaler[1]*, *Naomi A. Rose*[2], *A. Mel Cosentino*[3,4] and *Andrew J. Wright*[5,6]

[1] Blackbeard Biologic–Science and Environmental Advisors, St. Michaels, MD, United States, [2] Animal Welfare Institute, Washington, DC, United States, [3] Wild Earth Foundation, Puerto Pirámides, Argentina, [4] University of Strathclyde, Glasgow, United Kingdom, [5] Environmental Science and Policy, George Mason University, Fairfax, VA, United States, [6] Gateway Antarctica, University of Canterbury, Christchurch, New Zealand

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> "I wept, because I knew that this fleeting opportunity to bridge, no matter how tenuously, the ever-widening chasm that is isolating mankind from the totality of life, had perished in a welter of human stupidity and ignorance—some part of which was mine."
>
> A Whale for the Killing (Mowat, 1972).

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***Correspondence:**
Andrew D. Thaler
andrew@blackbeardbiologic.com

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INTRODUCTION

When Farley Mowat wrote *A Whale for the Killing* in 1972, the titular fin whale, stranded and intentionally wounded in a Newfoundland pond, was long dead, yet the story of Moby Joe and the spectacle surrounding her death would become a cornerstone of the emerging anti-whaling movement (see below). The media frenzy that descended on the small town of Burgeo as the whale struggled to survive, and the subsequent publication of Mowat's book, are among the first examples of efforts to turn spontaneous outpourings of outrage, curiosity, or empathy into conservation action by actively focusing media attention, a phenomenon that we have dubbed *moment inertia*. We use "moment" because this phenomenon arises from focus of attention around a single, clarifying event, or moment, and "inertia" because that attention propagates, undirected, through media unless acted upon by outside forces, much like physical inertia. Almost half a century later, the events leading to the publication of *A Whale for the Killing* stand among the most effective uses of moment inertia in the conservation movement.

The unnatural deaths of individual animals can draw attention to important conservation issues such as poaching, habitat destruction, and biodiversity loss. When public interest is piqued by moment inertia, strategic campaigning can transform that interest into action. Moment inertia is fleeting and channeling public attention toward achieving wider conservation goals requires a carefully planned response to capitalize on what may, at times, seem like superficial public engagement.

Understanding moment inertia, its limitations, and how it can be used to focus and enhance existing campaigns is key for effectively realizing conservation gains. Here, we examine three cases of moment inertia—one based on outrage, one based on curiosity, and one based on empathy—and present a strategic approach for transforming this moment inertia into conservation action.

CECIL THE LION

Public outrage is often the most visible and visceral form of moment inertia. In 2015, the killing of Cecil the Lion sparked massive outcry against his hunter, trophy hunting in general, and Zimbabwean wildlife management (Nelson et al., 2016, and see Beauchamp, 2015, for a summary of the public reaction). Intense media coverage galvanized symbolic actions by remote agencies and

stakeholders. While some animal welfare and wildlife conservation organizations used this event to solicit donations, in general conservation scientists and practitioners floundered. Some tried to redirect public interest toward bigger, albeit unrelated, issues, while others directed their scorn at those who were outraged, arguing that the public's attention was incorrectly focused on a marginal issue (see Howard, 2015, for an overview of the various reactions). These responses highlight a lack of understanding about the psychology of public outrage. Mass outrage responses are most often triggered by immaterially harmful acts—those with negligible long-term consequences—that permit moral signaling (Tannenbaum et al., 2011).

Examination of the timing of media events *post facto* shows that the initial news broke on both traditional and social media simultaneously. The initial growth phase that drove this moment inertia evolved over a 2-day period in which the story spread through multiple media markets from geographically diverse regions synchronously (Macdonald et al., 2016). This is a telltale sign of a concerted effort to generate "earned media," i.e., coverage gained though newsworthiness rather than paid advertising or via owned media (Thaler, personal observation). Organizations that effectively leveraged the moment inertia surrounding Cecil the Lion took advantage of the moral signaling inherent in the public outrage model to solicit donations, grow mailing lists, and pressure philanthropists. These are tactics that can facilitate longer, less event-dependent conservation campaigns while providing instant gratification to the outraged audience.

The specific timing or location of events such as Cecil's killing cannot be predicted and mobilizing quickly to achieve positive conservation outcomes can therefore be difficult. However, while these "outrage" events appear to be random, they are often also inevitable over the long-term, and the public response can be predictable. Poaching iconic animals, negative human-animal interactions, or human-induced disasters will invariably occur and can be anticipated. Quickly pairing such events with an appropriate pre-planned response could allow conservation professionals to utilize these moments for conservation gains.

THE (NON) EXPLODING WHALE OF NEWFOUNDLAND

Curiosity, especially morbid curiosity, can be a powerful motivator for harnessing moment inertia. When a blue whale stranded in a small, remote port in Newfoundland, Canada in 2014 (BBC, 2014), it received some local and regional coverage while various government agencies debated who had ultimate jurisdiction over its removal (Globe and Mail, 2014). Upwell, an ocean NGO focused on analyzing ocean messaging online, identified the incident as a flashpoint to generate moment inertia (Thaler, personal observation). Upwell formed a small campaign around the event, with the following goals: draw attention to the town, which was struggling with receiving government assistance to dispose of the corpse; call attention to protections for stranded marine mammals; provide a humorous resource to inform the public about marine mammal strandings; and significantly increase the online conversation about whale strandings using Upwell's Big Listening attention model (see Weidinger et al., 2013).

Upwell initiated a social media marketing plan, mobilized a highly-engaged mailing list, and launched HasTheWhaleExplodedYet.com (now expired), which provided visitors with continuous updates about the stranding event and resources on the appropriate treatment of stranded marine mammals, as well as contact information for local and regional stranding networks.

The resulting moment inertia grew throughout a week-long news cycle, driving nearly a million unique visitors per day to HasTheWhaleExplodedYet.com. Hundreds of articles about the Newfoundland whale and whale strandings were generated, as well as about the science and broader cultural associations of exploding whales (e.g., Bhatia, 2014; Goldman, 2014; Thaler, 2014). The week-long campaign reached its zenith when an exploding whale sketch was featured on the weekly sketch comedy show *Saturday Night Live* (Season 39, Episode 20). The whale never actually exploded and the Royal Ontario Museum sent a team to haul the carcass away for research (O'Connor and Bailey, 2014).

A Google trend analysis of searches for *Cecil lion* and *exploding whale* reveals the subtle differences between these two events and can help campaigners design strategies that complement these patterns. Prior to his killing, search traffic for *Cecil lion* was, understandably, 0% relative to maximum search volume. In the week following the killing, a significant attention spike was seen, with South Africa, Canada, the United States of America, the United Kingdom, Australia, France, and India responsible for the bulk of search volume. This spike quickly tapered to a long tail which persisted for 4 months at 1–2% of peak search volume before fading back to a 0% baseline (**Figure 1A**). From the period beginning 1 month after the event until May 8, 2017, the mean baseline for *Cecil lion* was 0.3% of the maximum, with a standard deviation of 0.67%.

Prior to the Newfoundland event, search traffic for *exploding whale* was 1.49% relative to the maximum search volume. (Note: a smaller spike in *exploding whale* searches occurred in 2013 surrounding an exploding sperm whale in the Faroe Islands, which was also, in part, the result of an Upwell campaign. We have calculated the baseline from May 13, 2012 to just before the Faroe event.) Searches were more localized to Canada, the United States of America, and the United Kingdom. From the period beginning 1 month after the event until May 5, 2017, the mean baseline for *exploding whale* was 2.21% of the maximum, with a standard deviation of 1.25%, indicating that baseline attention almost doubled following the Newfoundland incident (**Figure 1B**).

The goals of a campaign will determine whether the best outcome is a large, international attention spike with a relatively short baseline shift or a smaller, more regional attention spike with a relatively longer baseline shift.

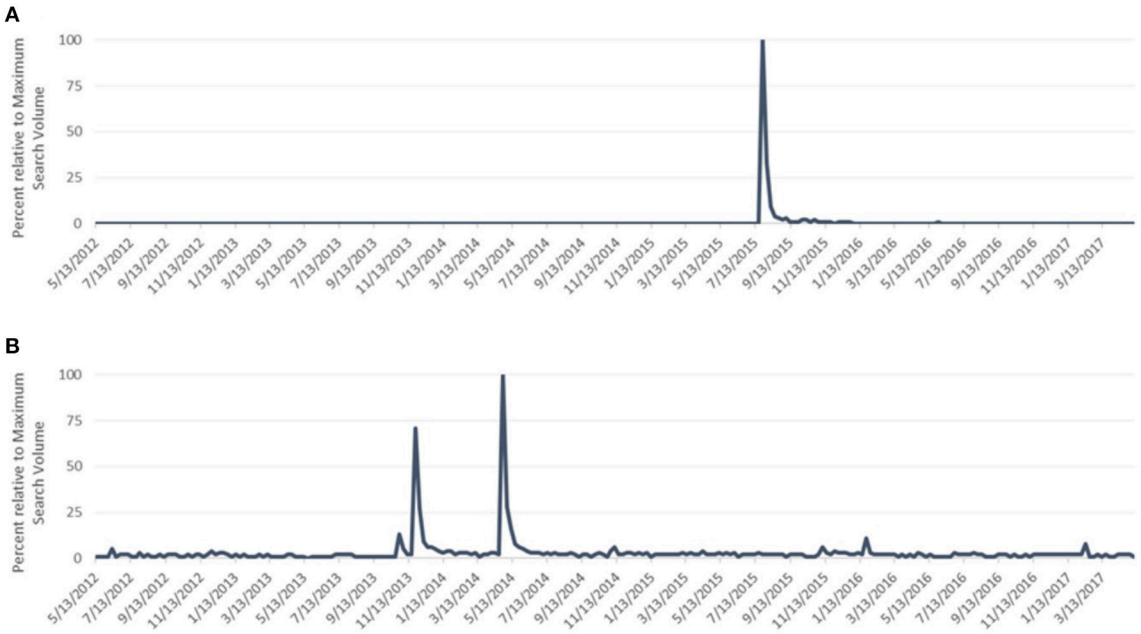


FIGURE 1 | Search volume for "Cecil lion" **(A)** and "exploding whale" **(B)** normalized against maximum search volume. Data provided by Google Trend analysis.

A WHALE FOR THE KILLING

Farley Mowat connected the public with another whale in Newfoundland; it was 1967 and the whale was very much alive. Moby Joe was a fin whale naturally trapped in a tidal pond near the town of Burgeo. The whale was shot at by hunters and other curious onlookers. Through a wave of press releases, articles, and radio interviews, Mowat established superficial protections for the trapped whale (Mackinnon, 2014), although these ultimately proved ineffective, as the whale eventually succumbed to her injuries. When Mowat later published *A Whale for the Killing*, detailing the event in his unique and uncompromising style, he not only connected his audience with the whale, but also the daily struggles of the people of Burgeo. This created a narrative that had no central villain and was empathetic even to those who caused the whale harm. The book became a cornerstone document in the emerging anti-whaling movement. That Mowat was working within the technological constraints of a less connected era serves to highlight that it is not the speed, reach, and breadth of the internet and social media, but rather preparation, tactical thinking, and a little bit of luck, that transforms moment inertia into effective conservation action.

HARNESSING AN INERTIAL MOMENT

Though these examples vary in scope, timing, and available technologies, they can inform strategies and tactics for mobilizing moment inertia. Ephemeral events can be used to leverage donations and grow audiences, but moment inertia often falls short of producing long-term behavioral changes. A strategic, well-crafted response can encourage short-term action from legislators and other decision makers that, when combined with a larger campaign, yields lasting consequences. Building a network of experts who can speak to both the specific context of an inertial event and the broader conservation issue makes it possible to quickly tailor and deploy strategic campaigns. Nurturing a community of experts in advance has proven critical in rapidly preparing and disseminating a response to specific events (Thaler and Shiffman, 2015).

Moment inertia is, in many cases, a product of dissociation—the primary audience is generally unfamiliar with the people and places associated with the event. Any effective conservation outcome necessarily affects the people geographically and culturally tied to an issue. Campaigns that fail to understand how those most directly connected relate to these animals and ecosystems are generally ineffective (Singleton, 2016), especially when there are issues surrounding traditional or economic use tied to the community. Local allies are essential to most conservation initiatives. To establish non-exploitive, local commitment to conservation goals, effective campaigns must address the cultural values of affected communities.

Particularly in outrage-based scenarios, there is a tendency to try and identify villains to which anger can be directed, but in all situations in which there is a perceived environmental injustice, the public seeks an antagonist. In many cases the larger context precludes placing the blame on a single person or group, even where there is a clear principal actor. This can lead to substantial challenges when it comes to harnessing moment inertia: the audience is looking for immediate gratification. The villain narrative can provide a clear, achievable goal, but it can also backfire. Focusing on a discrete villain obscures larger challenges and creates an additional barrier to achieving conservation goals.

Less intuitively, even in cases where blame can be placed, any attention generated from moment inertia is lost the instant "justice is served," and may ultimately result in a decline in public concern due to the impression that the problem is solved and no further action is required.

Conservation activism following moment inertia is a balancing act between strategic planning and a quick, tactical response. When the catalyst is moral outrage, it is important to allow people to be angry, rather than to try and curb such responses. In these circumstances, it is possible to leverage predictable moral signaling into tangible conservation gains.

Regardless of the emotional reaction—outrage, curiosity, or empathy—the general guidelines for conservationists leveraging moment inertia are the same. First, planning for pseudorandom events is essential to produce meaningful outcomes. Second, understanding the limitations of campaigning on an inertial moment will help establish and achieve concrete, realistic goals. Third, the call to action must be informed by the local context, address local cultural values, and be delivered by those who can connect with the public. Finally, it is critical to maintain a factual basis while acknowledging the emotions involved.

With foresight, a focus on concrete goals, and an understanding of the strengths and limitations inherent in moment inertia, these events can be harnessed to help achieve lasting conservation successes.

AUTHOR CONTRIBUTIONS

AT, NR, AMC, and AW participated in the Cecil the Lion panel at Fourth International Marine Conservation Congress (IMCC4) and subsequent discussions, which led to the production of this manuscript. AW drafted the initial manuscript. AT, NR, AMC, and AW contributed to revising and editing. AT performed social metric analyses.

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An Online Survey of Public Knowledge, Attitudes, and Perceptions Toward Whales and Dolphins, and Their Conservation

*Whitney Naylor and E. C. M. Parsons**

Department of Environmental Science and Policy, George Mason University, Fairfax, VA, United States

In 2015, an online survey was conducted to investigate public attitudes and perceptions toward key cetacean (whale, dolphin, and porpoise) conservation and "hot topic" issues such as legislative protection and whaling ($n = 858$). The vast majority of the participants in this study indicated their permanent residence was the United States ($n = 577$) or India ($n = 251$). Perceptions of participants on the conservation priority of cetacean species did not match with the factual IUCN status, where most participants assumed that the larger and more charismatic whales (blue whale, 24.01%; humpback whale, 22.14%; and killer whale, 23.43%) were more endangered or more important to conserve than the small cetacean species such as the Vaquita or Hector's dolphin. Additionally, 39.74% of participants indicated that they thought bottlenose dolphin was the most important to conserve. More members of the public highlighted non-existent (fake) species (e.g., pygmy short-fined whale, lump-headed dolphin, and majestic spotted dolphin) as being of conservation concern than certain species of actual, genuine concern. The majority of participants considered dolphins and whales to be "under protected" or only "slightly protected" (29.95%; 41.96%, respectively) and expressed that marine mammal conservation laws and policies were "very important" or "important" (47.43 and 37.88%, respectively). In addition, 86.83% of participants expressed opposition to the hunting of dolphins and whales (57.93% "strongly opposed" and 28.90% "opposed"); however, only 47.44% of participants were aware that several countries are still involved in whaling. A lack of awareness of the conservation status of whales and dolphins and continued whaling activities suggests that greater outreach to the public about the conservation status of whale and dolphin species is needed.

Keywords: public opinion, dolphin, conservation, cetaceans, public attitudes, public awareness, whaling, public knowledge

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Edited by:
Luis Cardona,
Universitat de Barcelona, Spain

Reviewed by:
Danielle Kreb,
RASI Conservation Foundation, Indonesia
Manel Gazo,
Universitat de Barcelona, Spain

****Correspondence:***
E. C. M. Parsons
ecm-parsons@earthlink.net

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INTRODUCTION

Attitudes toward marine conservation can vary considerably depending upon country of residence and what issue is being discussed. Eagly (1992) defined attitude as "a tendency or state internal to a person which biases or predisposes a person toward evaluative responses which are to some degree favorable or unfavorable" (pp. 694). A person's attitude is characteristic of his or her evaluation of the representative object (Ajzen and Fishbein, 1977). Furthermore, Rosenberg (1956) stated that "a

strong and stable positive affect toward a given object should be associated with beliefs to the affect that the attitude object tends to facilitate the attainment of a number of important values, while strong negative affect should be associated with beliefs to the effect that the attitude object tends to block the attainment of important values" (pp. 367).

Several different variables can affect the attitudes people have toward the natural environment including gender, locality, ethnicity, age, income, and wildlife activities. Kellert (1976) found that people's attitudes significantly change with age. Attitudes of children, ranging in ages from 6 to 9, tend to focus around emotional relationships toward animals. They then shift to cognitive or factual attitudes upon becoming a young adolescent (ages 10–13) and then shift again in post-adolescence to attitudes encompassing ethical concern and ecological awareness of the role of animals in their natural habitats (Kellert, 1976). Kellert and Berry (1980) found significant differences between male and female attitudes, knowledge, and behavior toward animals, concluding that gender lies among the most important demographic factors that influence attitudes about animals. The results showed that males tended to value animals for practical and recreational reasons, in contrast to females who were more inclined to express concern for the consumptive exploitation of wildlife and value animals as objects of affection (Kellert and Berry, 1980).

Overall, education is the factor found most likely to change people's attitudes and perceptions (Kellert, 1996; Thompson and Mintzes, 2002). A direct link was also found between an individual's education level and that individual's level of interest, awareness, and concern for environmental issues (Kellert and Berry, 1980; Thompson and Mintzes, 2002). A study by Barney et al. (2005) looked at the effects of education level on knowledge, attitude, and harassment behavior toward bottlenose dolphins and the effects of knowledge structure and attitude on harassment behavior. The study used concept maps to measure the level and complexity of knowledge each individual had regarding dolphins and then had individuals respond to a Likert-type attitudinal inventory to assess their attitudes toward dolphins. Respondents for the tests were students in grades 5, 8, and 11 along with first year university students in general psychology, third year university students in marine biology, and graduate students in the MS program in marine biology. Furthermore, Barney et al. (2005) found that knowledge of dolphins increased with education level and that with increased knowledge, individuals were more likely to have an environmentally friendly attitude toward dolphins and less likely to have a negative attitude. Similar findings were made in studies looking at public knowledge, attitude, and behaviors toward sharks (Thompson and Mintzes, 2002; O'Bryhim and Parsons, 2015).

Thompson and Mintzes (2002) also used concept maps to judge knowledge in a comparable group of students and found that knowledge did increase with grade level. The study concluded an overall positive correlation between scientific and naturalistic[1] attitudes and knowledge complexity, in contrast to a negative correlation between utilitarian[2] and negative attitudes and knowledge complexity (Thompson and Mintzes, 2002). It should be noted that almost all of the college level students that participated in either study were either taking a marine biology class or were in a marine biology MS graduate study program and had received information regarding these animals prior to participation in the study, which could insert a bias in the results (Thompson and Mintzes, 2002; Barney et al., 2005).

The attitude a person holds toward any animal seeking protection is important because it can affect their behavior toward those animals (Thompson and Mintzes, 2002). Kraus (1995) stated that attitudes in some fashion can guide, influence, direct, shape, or predict a person's behavior. Furthermore, perceptions of a particular species can be influenced by demographic characteristics, fear of, and empathy toward an animal (Kellert, 1985). Karaffa et al. (2012) asked whether the name used for an animal influenced respondents' opinions regarding its conservation and found that on average species names with negative connotations prompted less support for conservation compared to charismatic sounding names which roused greater support for conservation. They suggest alternative names might make social marketing campaigns more effective and that renaming a species could be a more cost effective way to enhance conservation support (Karaffa et al., 2012).

While many of these species are threatened or endangered, some are also used to focus concern and awareness on less visible problems of ecosystem degradation (Barney et al., 2005). Dolphins and many other marine mammals typically generate positive and aesthetic public attitudes (Kellert, 1999); much of this can be credited to mass media outlets including films, television, and articles in magazines and newspapers (Barney et al., 2005). However, studies have shown a lack of awareness of other high profile threatened species such as whales, penguins, and polar bears (Scott and Parsons, 2004, 2005; Howard and Parsons, 2006; Parsons et al., 2010; Sitar-Gonzales and Parsons, 2012; Luksenburg and Parsons, 2013). Unfortunately, scientists suggest that these attitudes often encourage human-animal encounters that can be harmful or even fatal to the dolphins (Barney et al., 2005). Barney et al. (2005) found that more knowledgeable and environmentally responsible individuals were the least likely to participate in potentially disruptive or harmful harassment behavior toward dolphins.

This study differs from those studies described above as it addresses a more global audience instead of participants in a particular location.

The purpose of this study is to:

- Assess the participants' opinion on dolphin and whale conservation issues, whaling, and captivity;
- Determine if there was a significant relationship between knowledge, attitude, and behaviors as they pertain to dolphins and whales;
- Assess the participant's usage of social media and opinion of usefulness/accuracy for dolphin and whale information;

[1]Interest in direct experience with animals and the exploration of nature (Thompson and Mintzes, 2002).

[2]Concern for the environment as a system; for inter-relationships between wildlife species and natural habitats (Thompson and Mintzes, 2002).

- Explore the effects of gender and education level on a participant's behavior, attitude, and knowledge.

METHODS

The distribution of the survey instrument (see Appendix in Supplementary Material) used in this study was conducted using Amazon Mechanical Turk (MTurk). MTurk boasts a large and diverse workforce of over 100,000 from over 100 countries who complete thousands of tasks daily (Buhrmester et al., 2011). The site "brings together the people and tools that enable task creation, labor recruitment, compensation, and data collection" (Buhrmester et al., 2011). Within the MTurk site, individuals have the option to register as task creators (requesters) or paid task completers (workers). Task creators can create and post surveys, experiments, writing, etc. using technical scripts or simple templates or linking workers to external online survey tools (Buhrmester et al., 2011). Task completers can select available tasks and are subsequently paid upon successful completion of a task.

Surveys were available on MTurk from March to April 2015 to any MTurk task completer that had 1,000 or more Human Intelligence Task's (HITs) approved. No specific individuals were sought out for questioning, and individuals were asked to indicate whether they agreed or declined to participate in the survey. If they declined, the task completer was unable to view the survey questions or submit the survey. The survey instrument in this study as well as the procedure for its distribution was in accordance to the requirements and guidelines of the Human Subjects Review Board at George Mason University, and was approved by this body.

The response rate to the survey was $n = 1{,}020$ individuals and an incentive of \$0.25 USD was offered to each task completer if the survey was approved. Buhrmester et al. (2011) found that participation in surveys on MTurk is affected by compensation rate and task length; however, participants can still be recruited rapidly and inexpensively. Realistic compensation rates after successfully completing a survey do not affect data quality, and the data obtained from a survey conducted on MTurk is at least as reliable as the data obtained from traditional methods (Buhrmester et al., 2011). Some surveys that were submitted were not filled out in their entirety ($n = 162$) and information that was provided was not used in this study. For the purpose of this study, 858 completed surveys were utilized. This study is primarily an investigation of attitudes and perceptions in the United States and India, but responses and differences between these two countries are analyzed and explained. An MTurk-based study conducted by Ross et al. (2010) found that over half of survey participants were from the United States and approximately 1/3rd were from India indicating an increase in MTurk international users. Task completers from the United States are offered the option of dispersing earnings from surveys to an Amazon gift card or to a personal bank account; task completers in India also have the option to transfer earnings in Indian Rupees to personal bank accounts. All other international task completers can only transfer earnings into Amazon gift card.

A variety of topics are covered in this survey including conservation, captivity, whaling, and social media, with the number of questions devoted to each issue varying considerably. For many questions, no distinction was made among dolphin and whale species. Twelve attitude questions were included in the survey. Six additional questions focused on knowledge of whales and dolphins. Six behavioral questions were also included on environmental organization membership, subscription to environmental/scientific/animal magazines, participation in whale watching, watching animal programs, preference on observing whales and dolphins, and travel to countries involved in whaling. Demographic questions elicited information on respondent's age, place of residence, occupation, education, and gender.

The program SPSS was used for all statistical analysis of the data.

The survey questions were partitioned into attitude, behavior, and knowledge categories in order to create indexes that would measure a person's knowledge level, attitude, and behavior toward dolphins and dolphin conservation. An index was not created for the demographic and social media questions.

The knowledge index consists of three questions (survey questions 2, 16, and 18) that were each coded into binary inputs. These questions did not have a logical hierarchy, and therefore could not be put into a sequential number input. Question 2 asked which dolphin/whale species listed was the most important to conserve including five known dolphin/whale species, three fake species, one non-cetacean species, and the category "not applicable." The five known dolphin/whale species were coded as one and all other choices as zero. To create the knowledge index these questions were added together to give a number out of three, with higher scores representing more dolphin/whale knowledge.

The attitudinal index is made up of 12 questions (survey questions 1, 14, 15A-H, 19, and 20), and used to judge a participant's attitudes toward dolphins and dolphin conservation. Question 1 asked how important participants thought dolphin/whale conservation was from very important, important, slightly important, slightly unimportant, unimportant, to very unimportant. This question was scored as a Likert item (scale using levels of agreement or disagreement) with very urgent being worth three and not at all urgent being worth zero. Question 20 was also scored in the same manner. Questions 14 and 19 were scored with an answer of strongly oppose worth two and strongly support as zero. To create the attitude index these questions were added together to give a number out of 18, with higher scores representing a more pro-dolphin/whale attitude.

The behavior index is made up of six questions (survey questions 4, 5, 6, 13, 17, and 27) and used to measure a participant's general behavior toward dolphins. Question 13 asked in which way the individuals would prefer to see dolphin/whales if costs were similar and all options available in a given location. This question was not coded into binary. A selection of dolphin/whales in the ocean from a land-based vantage point received a two, on a dolphin/whale watching boat

trip a one, and in a marine park or aquarium a zero. The sum of these questions was used to create an index with a scale of zero to seven with higher scores representing a more pro-dolphin/whale behavior.

As previously mentioned, the demographic questions (survey questions 22 through 26) from the survey were not used to create another index, but rather were used separately for analyses.

A bivariate correlation was used to test for significant relationships between the knowledge, attitude, and behavior indexes. Univariate analysis of variance (ANOVA) tests were used to determine if there was a significant difference between level of education and knowledge, attitude, and behavior indexes. Independent *T*-tests were also used to see if the mean attitude, knowledge, and behavior indexes of females differed from that of the males who took the survey and if the mean attitude, knowledge, and behavior indexes of participants with a social media account(s) differed from that of participants without a social media account.

RESULTS

Demographics

The majority of the participants were male (60.96%; $n = 523$), with females making up the remaining 39.04% ($n = 335$). The age range of the sample population was from 19 to 73 years of age with a mean age of 35 years ($SD = 10.9$) (**Figure 1**).

The majority of the participants in this study indicated their permanent residence was the United States ($n = 577$). Approximately 1/4th of the participants indicated their permanent residence was in India ($n = 251$) and the remaining participants indicated that their permanent residence was in Canada ($n = 3$), Croatia ($n = 2$), United Kingdom ($n = 2$), France ($n = 1$), Greece ($n = 1$), Lithuania ($n = 1$), Macedonia ($n = 2$), Mexico ($n = 1$), New Zealand ($n = 1$), Nigeria ($n = 1$), Pakistan ($n = 1$), Poland ($n = 1$), Romania ($n = 3$), Serbia ($n = 1$), Slovenia ($n = 1$), South Africa ($n = 1$), Sri Lanka ($n = 1$), Taiwan ($n = 1$), and Thailand ($n = 1$) (**Figure 2**).

In terms of education, approximately half of the participants ($n = 441$) had completed at least an associate's degree, while the remaining half was divided between participants who obtained a higher degree (Masters, Ph.D., M.D, or other terminal degree) (18.53%; $n = 159$) and participants who had up to some college but no degree (30.07%; $n = 258$) (**Table 1**).

Knowledge, Attitude, and Behavior

The knowledge index had a minimum possible score of zero and a maximum possible score of three. The participant's scores ranged from zero to three and the mean score was 1.169 ($SD = 0.878$). Higher index scores reflected a higher level of knowledge about dolphins and whales and related conservation issues (**Figure 3**).

An independent *t*-test was conducted to determine if the mean knowledge level of males differed from the mean knowledge index of females. Data were gathered from samples of 335 females and 523 males, with a female sample mean of 1.12 ($SD = 0.845$) and a male sample mean of 1.20 ($SD = 0.845$). The independent *t*-test indicated that the knowledge level means were not significantly different for males and females ($t = -1.405$, $df = 856$, $p = 0.160$).

An independent *t*-test was also conducted to determine if the mean knowledge level of participants with a social media account(s) differed from the mean knowledge level of those without a social media account. Data were gathered from samples of 700 participants with a social media account(s) and 158 without, with a social media account(s) mean of 1.21 ($SD = 0.881$) and a without social media account sample mean of 0.98 ($SD = 0.841$). The independent *t*-test indicated that the knowledge level means were significantly different for participants with a social media account(s) and without a social media account ($t = -3.084$, $df = 241.195$, $p = 0.002$). Thus, those participants with a social media account(s) were more likely to report higher knowledge of dolphins and related conservation issues.

A one-way ANOVA was conducted to determine if the mean knowledge level differed by level of education of the participants. The one-way ANOVA was found to not be statistically significant [$F_{(4, 858)} = 0.624$, $p = 0.646$, $\eta^2 = 0.003$].

The attitude index had a minimum possible score of zero and a maximum possible score of 18. Higher scores for this index represented the likelihood that a participant's attitudes would be more dolphin and whale positive and conservation oriented. The participant's scores ranged from one to 18 and the mean score was 10.713 ($SD = 3.284$) (**Figure 4**).

An independent *t*-test was conducted to determine if the mean attitude level of males differed from the mean attitude level of females. Data were gathered from samples of 335 females and 523 males, with a female sample mean of 11.28 ($SD = 3.199$) and a male sample mean of 10.53 ($SD = 3.290$). The independent *t*-test indicated that the attitude level means were significantly different for males and females ($t = 4.065$, $df = 856$, $p < 0.0001$). Thus, there is a significant difference in attitudes between genders, with females being significantly more positive toward cetaceans than males.

A one-way ANOVA was conducted to determine if the mean attitude level differed by level of education of the participants. The one-way ANOVA was found to be statistically significant [$F_{(4,858)} = 4.171$, $p = 0.002$, $\eta^2 = 0.019$]. Those participants with Master's Degree were significantly more positive toward cetaceans than a combined group of participants with high school diploma/GED and some college, but no degree.

The behavior index had a minimum possible score of zero and a maximum possible score of seven. The participant's scores ranged from zero to seven and the mean score was 2.334 ($SD = 1.189$) Higher index scores reflected more pro-dolphin/whale and pro-conservation behavior (**Figure 5**).

An independent *t*-test was conducted to determine if the mean behavior level of males differed from the mean behavior level of females. Data were gathered from samples of 335 females and 523 males, with a female sample mean of 2.51 ($SD = 1.176$) and a male sample mean of 2.22 ($SD = 1.184$). The independent *t*-test indicated that the behavior level means were significantly different for males and females ($t = 3.553$, $df = 856$, $p < 0.000$). Therefore, female participants more likely to report positive behaviors toward dolphins and dolphin conservation than male participants.

A one-way ANOVA was conducted to determine if the mean behavior level differed by level of education of the participants.

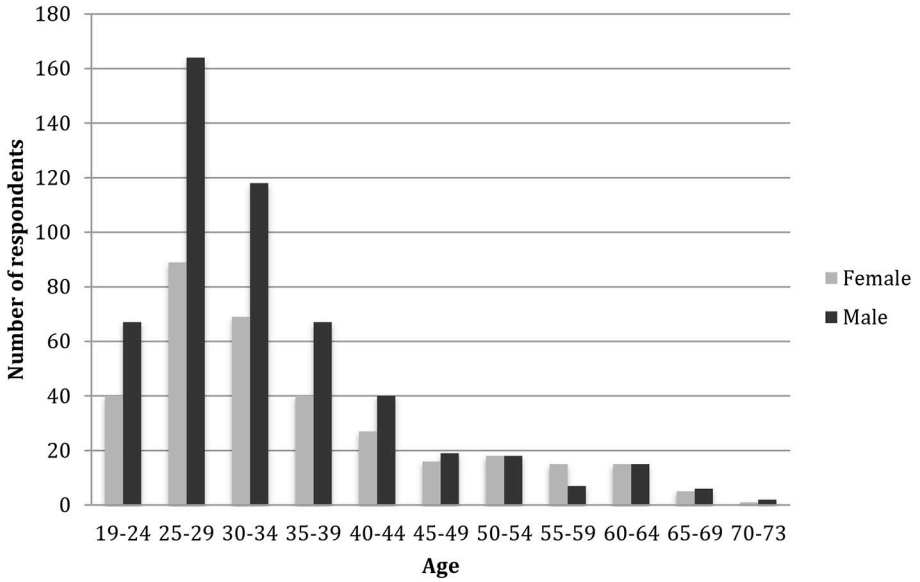


FIGURE 1 | Distribution of ages, separated by males and females, of survey participants ($n = 858$).

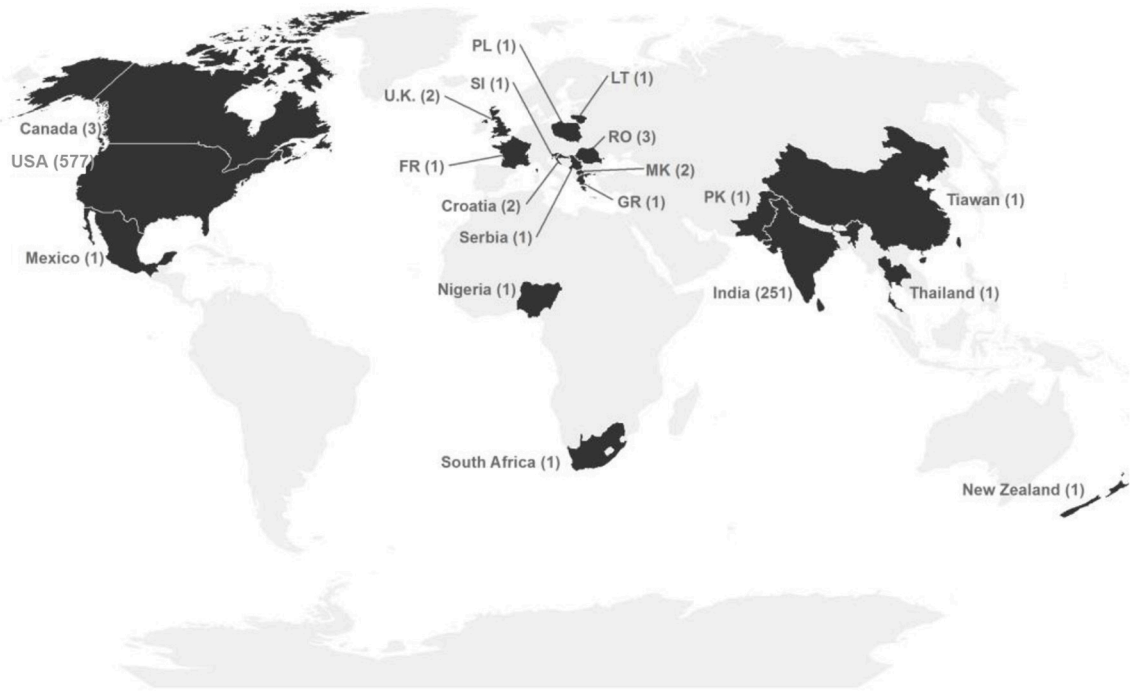


FIGURE 2 | Survey participants' country of permanent residence. The number of participants per country is in parentheses ($n = 854$).

Unlike with the attitude index above, the one-way ANOVA was found to not be statistically significant for the behavior index [$F_{(4,858)} = 1.363$, $p = 0.245$, $\eta^2 = 0.006$].

A Pearson correlation coefficient was computed to determine if there is a relationship between knowledge, attitude, and behavior levels. The null hypothesis that the correlation is 0 was rejected at the 0.01 level of significance. Therefore, there is a positive correlation between knowledge level, attitude level, and behavior level (**Table 2**). Therefore, higher levels of knowledge was linked to more positive attitudes toward cetaceans and the conducting of more cetacean-positive behaviors.

TABLE 1 | Responses to the question "Please indicate the highest level of education that you have completed."

Education level	Less than high school	High school diploma/GED	Some college, no degree	Associate/ Bachelor degree	Master's degree	Ph.D., M.D., or other terminal degree
Percentage (%) of respondents	0.70	7.93	21.44	51.40	17.37	1.16

Answers are represented as percentages (n = 858).

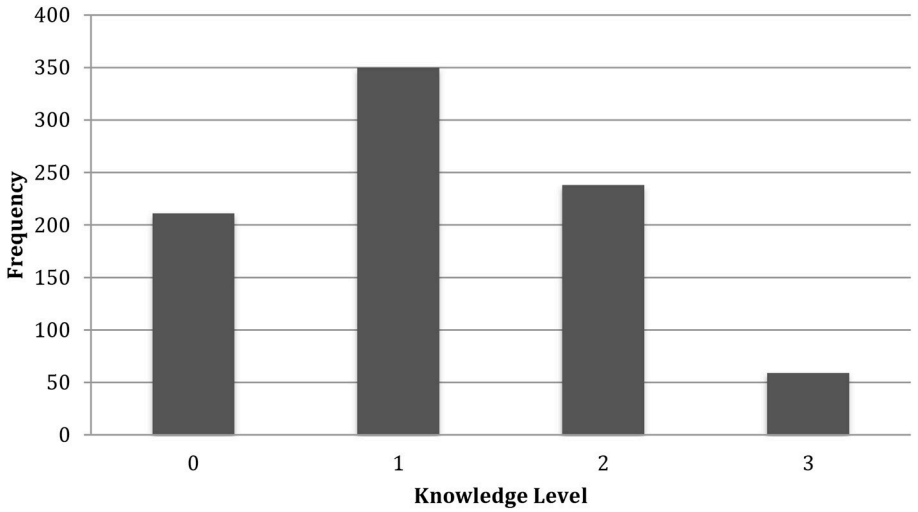


FIGURE 3 | Range of respondents' knowledge levels, indicating how much or little they know about dolphins and whales. Higher scores indicate more knowledge (*n* = 858).

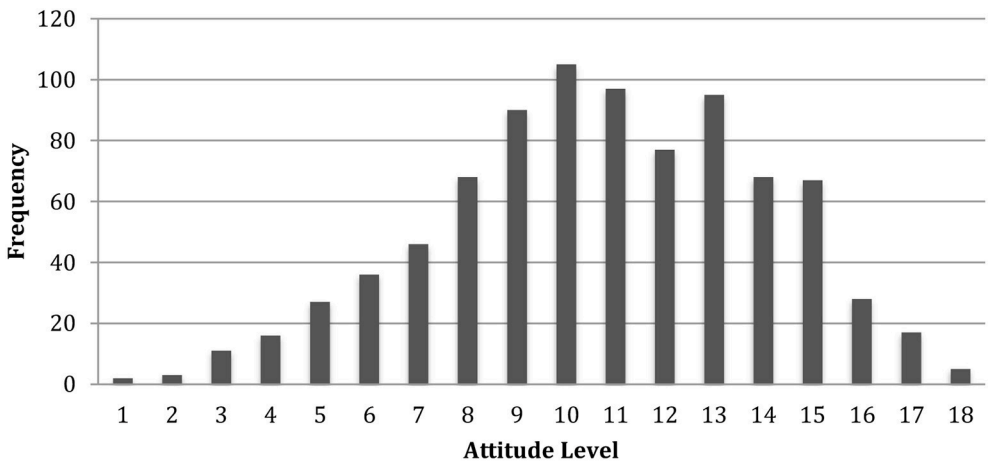


FIGURE 4 | Range of respondents' attitude levels, indicating how pro-dolphin they are. Higher scores indicate a more pro-dolphin attitude (*n* = 858).

Conservation

Survey participants were asked how important they felt whale and dolphin conservation was. The participant's responses are noteworthy because approximately 95% of participants felt conservation was important at some level, with only 4% stating that it was "unimportant" to them (**Table 3**). Similarly, 96.41% of participants from the US and 95.49% of participants from India felt whale and dolphin conservation was important at some level (**Table 4**). Participants were then asked which dolphin/whale species from a given list was the most important to conserve (**Figure 6**). **Figure 7** displays which dolphin/whale species from a given list participants from the United States and India felt was the most important to conserve. **Table 5** displays the IUCN Red List and US Endangered Species Act

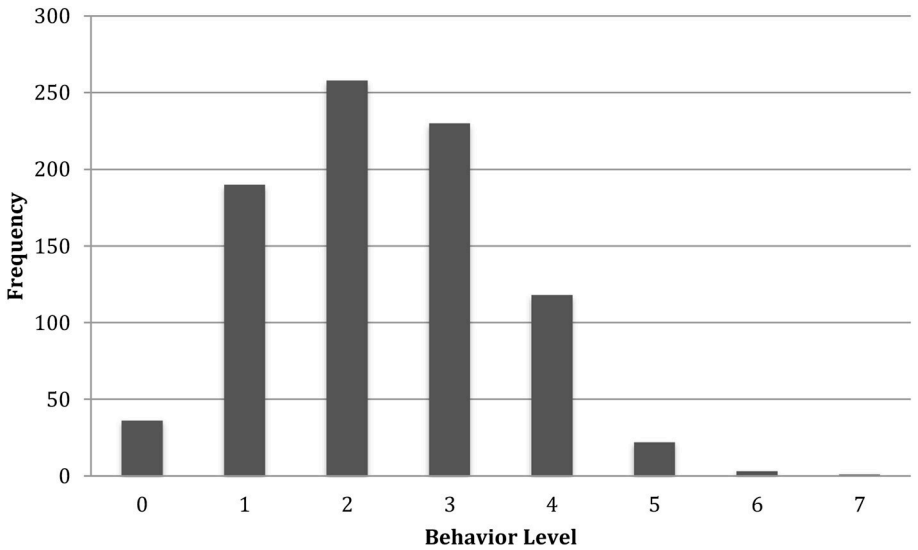


FIGURE 5 | Range of respondents' behavior levels, indicating how pro- or anti-dolphin they would behave. Higher scores indicate more pro-dolphin behavior ($n = 858$).

TABLE 2 | Correlation between previously recorded knowledge, attitude, and behavior levels.

		Knowledge	Attitude	Behavior
Knowledge	Pearson correlation	–	–	–
	N			
Attitude	Pearson correlation	0.183**	–	–
	N	858		
Behavior	Pearson correlation	0.144**	0.260**	–
	N	858	858	

***Correlations are significant at the 0.01 level (two-tailed).*

(ESA) listing for each dolphin and whale species respondents were asked to choose from. Of note, approximately 40% of respondents felt the bottlenose dolphin was the most important species to conserve. Over 40% of participants from the US thought the bottlenose dolphin was the most important to conserve compared to only 31% of participants from India. Despite essentially being the same species, the "Vaquita porpoise" received more votes than the [Gulf of] California harbor porpoise, 3.61 and 2.68% respectively, but both species were highlighted by more than an order of magnitude less members of the public than noted the bottlenose dolphin as the chief species of concern. Additionally, the three "fake" species (the "Lump-headed dolphin"−6.17%, the "Majestic spotted dolphin"−7.0%, and the "Pygmy short-finned whale"−4.2%) received more votes collectively than the "California harbor porpoise," the "Vaquita porpoise," the "Northern right whale"−5.01%, and the "Fin whale"−4.9% (17.37 and 16.02%, respectively). Although not a cetacean, 16% of respondents felt the whale shark was the most important dolphin/whale species to conserve. Of note, significantly more participants from India felt the whale shark was the most important to conserve than participants from the United States.

Participants were also asked to indicate which species listed in the survey was the most endangered (**Figure 8**). **Table 6** displays the IUCN Red List and US Endangered Species Act (ESA) listing for each dolphin and whale species respondents were asked to choose from. Approximately 75% of respondents felt that the blue whale, the humpback whale, and the killer whale were the most important to conserve (24.01, 22.14, and 23.43%, respectively). Of note, significantly more participants from the United States felt the humpback whale was the most endangered and more participants from India felt the blue whale was the most endangered (**Figure 9**).

Less than 5% of respondents thought that marine mammal conservation laws and policies were unimportant (**Table 7**). Over one-quarter of the respondents felt that dolphin and whale species were under protected and another two-fifths indicated that dolphin and whale species are only slightly protected (**Table 8**). Of note, 37.27% of US participants indicated that dolphin and whale species were under protected compared to only 14.34% of participants from India (**Table 9**).

A chi-square test of association was conducted to determine if there was a relationship between how important participants felt dolphin and whale conservation was and the which way they would prefer to see dolphins and whales. Sixty percent of the participants prefer to see dolphins and whales on a dolphin/whale-watching boat trip. Participants that feel dolphin/whale conservation is "slightly unimportant" to "very unimportant" have a higher percentage of this preference, while the individuals that felt dolphin and whale conservation was very important had a higher preference to view dolphins and whales in the ocean from a land-based vantage point. However, there is not

TABLE 3 | Percentage of respondents whose opinion it was that dolphin/whale conservation was very important, important, slightly important, slightly unimportant, unimportant, or very unimportant ($n = 858$).

	Very important	Important	Slightly important	Slightly unimportant	Unimportant	Very unimportant
Percentage (%) of respondents	49.18	32.87	13.76	2.21	0.70	1.28

TABLE 4 | Percentage of respondents from the United States and India whose opinion it was that dolphin/whale conservation was very important, important, slightly important, slightly unimportant, unimportant, or very unimportant (US, $n = 577$; India, $n = 251$).

Percentage (%) of respondents	Very important	Important	Slightly important	Slightly unimportant	Unimportant	Very unimportant
United States	54.18	30.68	11.55	1.99	0.4	1.20
India	47.66	33.45	14.38	2.43	0.69	1.39

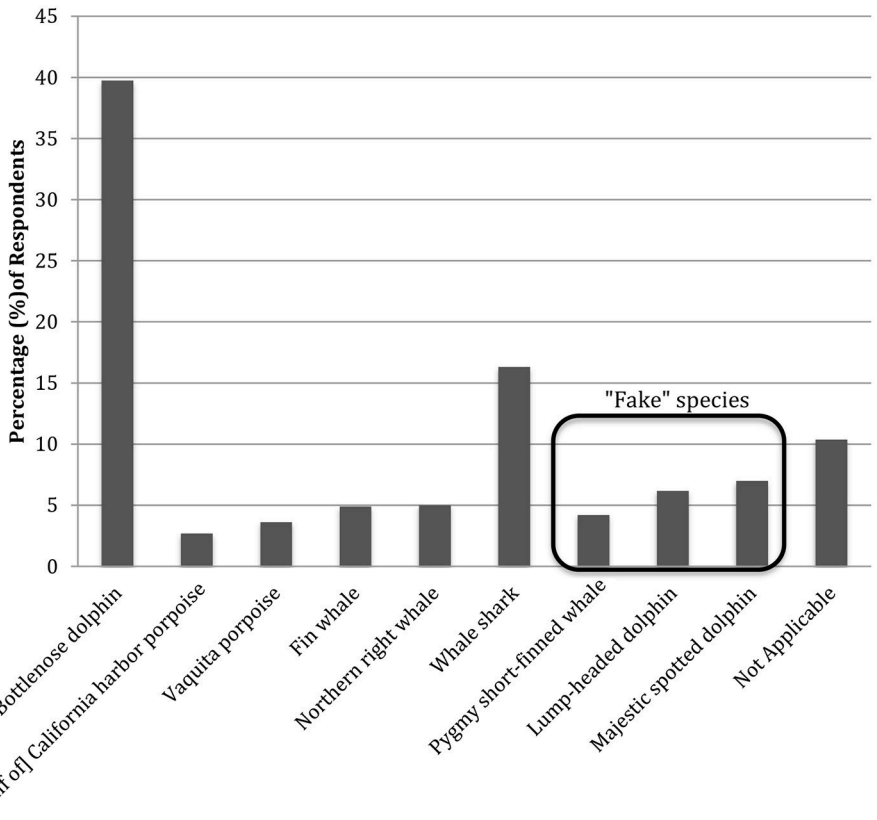


FIGURE 6 | Which dolphin/whale species listed respondents felt was the most important to conserve. The whale shark is not a cetacean species. The Pygmy short-fined whale, the Lump-headed dolphin, and the Majestic spotted dolphin were included as "fake" species ($n = 858$).

a significant association between how important participants felt dolphin and whale conservation was and which way they would prefer to see dolphins and whales ($x^2 = 7.144$, $df = 6$, $p = 0.308$).

A chi-square test of association was also conducted to determine if there was a relationship between how important participants felt dolphin and whale conservation was and if they were a member of any type of conservation/environmental group. There appears to be an association or relationship between the level of dolphin and whale conservation importance and membership in a conservation/environmental group ($x^2 = 9.693$, $df = 3$, $p = 0.021$). Examination of the standardized residuals suggests that respondents who feel dolphin and whale conservation is very important are significantly more likely to be a member of any type of conservation/environmental group (standard residual = 2.2) as compared to all other respondents.

A chi-square goodness-of-fit test was conducted to determine if the sample proportions were the same for how important the respondents felt marine mammal conservation laws and policies were. The study found a statistically significant

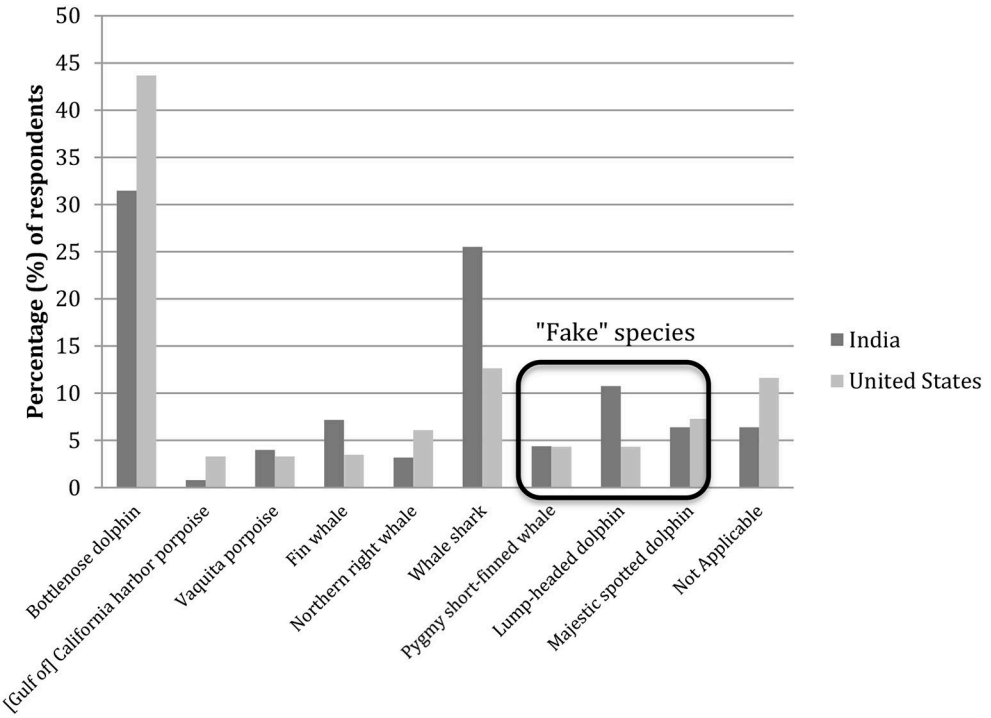


FIGURE 7 | Which dolphin/whale species listed respondents from the United States and India felt was the most important to conserve. The whale shark is not a cetacean species. The Pygmy short-fined whale, the Lump-headed dolphin, and the Majestic spotted dolphin were included as "fake" species (US, $n = 577$; India, $n = 251$).

TABLE 5 | Percentage of respondents whom felt the listed dolphin/whale species was the most important to conserve along with the IUCN Red List and US Endangered Species Act (ESA) listing for each species ($n = 858$).

Species	IUCN Red List	ESA listing	Percentage (%) of respondents
Bottlenose dolphin *Tursiops truncatus*	Least concern	Candidate species—Fiordland population	39.7
[Gulf of] California harbor porpoise (alternative name for the Vaquita porpoise)			2.7
Vaquita porpoise *Phocoena sinus*	Critically endangered	Endangered—throughout its range	3.6
Fin whale *Balaenoptera physalus*	Endangered	Endangered—throughout its range	4.9
Northern right whale (North Atlantic right whale) *Eubalaena glacialis*	Endangered	Endangered—throughout its range	5.0
Whale shark* *Rhincodon typus*	Vulnerable	Petition to list as either threatened or endangered	16.3
Pygmy short-finned whale**	"Fake species"		4.2
Lump-headed dolphin**	"Fake species"		6.2
Majestic spotted dolphin**	"Fake species"		7.0
Not applicable			10.4

**Not a cetacean.*
***Fake species name.*

difference between the levels of importance for marine mammal conservation laws and policies ($x^2 = 450.485$, $df = 3$, $p \leq 0.001$).

Survey Participant Behavior

Table 10 displays responses to four questions pertaining to participant's environmental related activities. Only 14% of respondents indicated they had ever gone on a dolphin/whale research expedition or a dolphin/whale-watching trip. Approximately half of the respondents indicated they watch animal programs on channels such as Discovery, Animal Planet, BBC, or National Geographic. However, only 6% of respondents indicated they subscribed to any environmental/scientific/animal

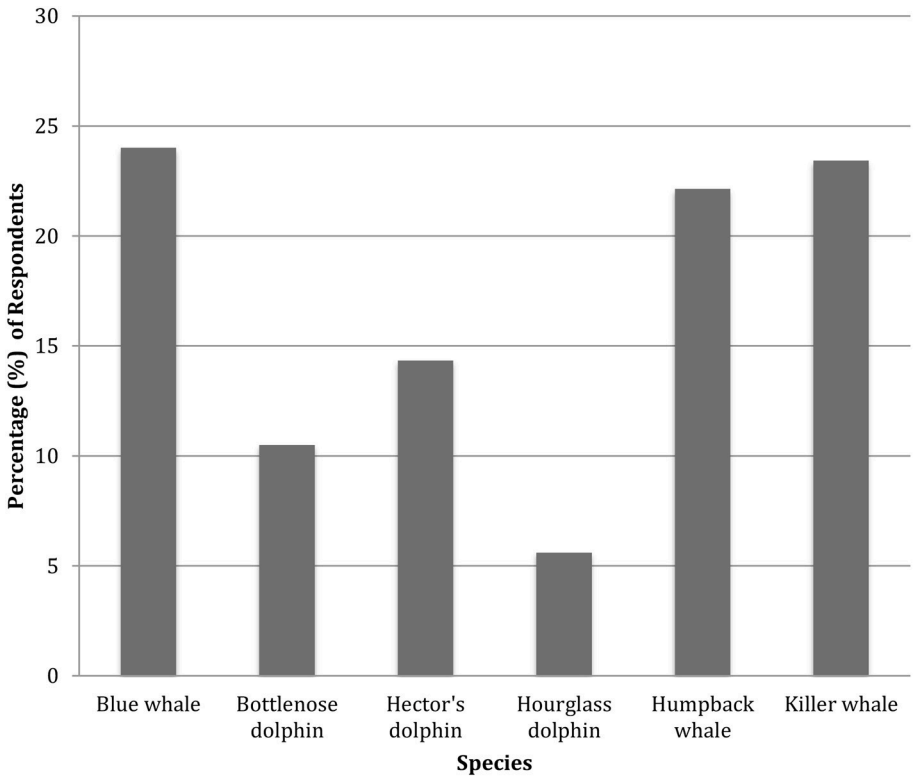


FIGURE 8 | Which dolphin/whale species listed respondents felt was the most endangered ($n = 858$).

TABLE 6 | Percentage of respondents whom felt the listed dolphin/whale species was the most endangered along with the IUCN Red List and US Endangered Species Act (ESA) listing for each species ($n = 858$).

Species	IUCN Red List	US Endangered Species Act (ESA) listing	Percentage of respondents
Blue whale *Balaenoptera musculus*	Endangered	Endangered—throughout its range	24.0
Bottlenose dolphin *Tursiops truncatus*	Least Concern	Candidate species—Fiordland population	10.5
Hector's dolphin *Cephalorhynchus hectori*	Endangered	Candidate Species—throughout its range	14.3
Hourglass dolphin *Lagenorhynchus cruciger*	Least concern	Not listed	5.6
Humpback whale *Megaptera novaeangliae*	Least concern	ESA Proposed Threatened-–2 distinct population segments: Central America DPS—Western North Pacific DPS ESA Proposed Endangered-–2 DPS Arabian Sea DPS—Cape Verde Island/Northwest Africa DPS ESA Endangered—throughout its range	22.1
Killer whale (Orca) *Orcinus orca*	Data deficient	ESA Endangered—Southern Resident killer whales (J, K, and L pods)	23.4

magazine. Of those participants that subscribed, 33% indicated they received National Geographic Magazine. An even lower number of participants, approximately 4%, stated that they were a member of any type of conservation/environmental group.

Whaling

This section considers participants attitudes toward whaling. Slightly less than half of respondents indicated that they were aware that several countries are still involved in whaling

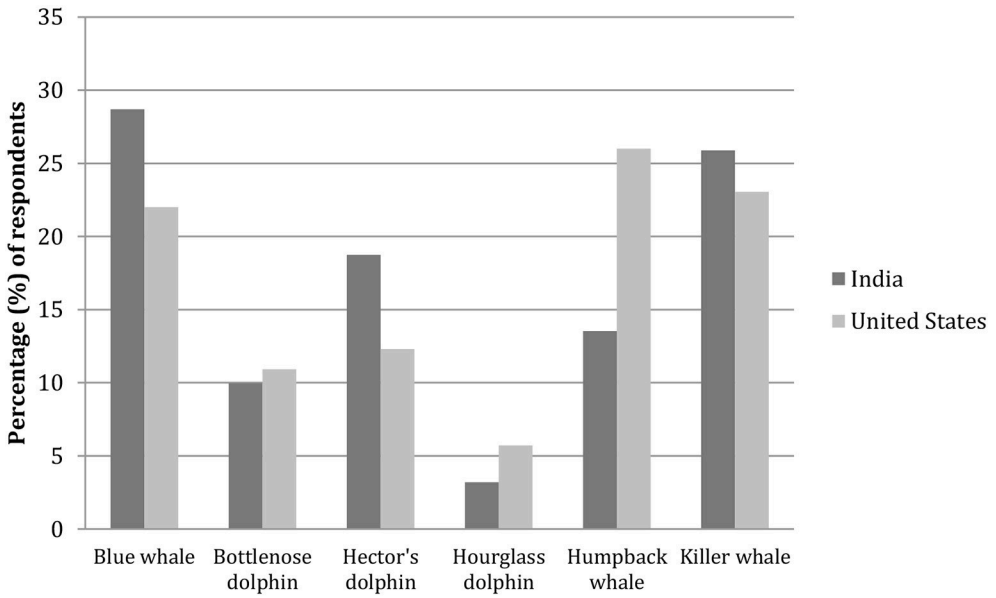


FIGURE 9 | Displays which dolphin/whale species listed respondents from the United States and India felt was the most endangered (US, $n = 577$; India, $n = 251$).

TABLE 7 | Responses to the question "How important do you think marine mammal conservation laws and policies are?"

	Very important	Important	Slightly important	Slightly unimportant	Unimportant	Very unimportant
Percentage (%) of respondents	47.43	37.88	10.50	2.56	0.70	0.93

Answers are represented as percentages (n = 858).

TABLE 8 | Responses to the question "How well do you think dolphin and whale species are protected worldwide?"

	Overprotected	Protected	Slightly protected	Under protected
Percentage (%) of respondents	3.61	24.48	41.96	29.95

Answers are represented as percentages (n = 858).

TABLE 9 | Responses from the United States and India to the question "How well do you think dolphin and whale species are protected worldwide?"

Percentage (%) of respondents	Overprotected	Protected	Slightly protected	Under protected
United States	1.21	15.42	46.10	37.27
India	9.16	46.22	30.28	14.34

Answers are represented as percentages (US, n = 577; India, n = 251).

TABLE 10 | Responses to four questions pertaining to participant's environmental related activities.

Question	Yes	No	*n*
Have you ever gone on a dolphin/whale research expedition or a dolphin/whale watching trip?	14.7	85.3	858
Do you watch animal programs on channels like Discovery, Animal Planet, BBC, or NatGeo?	48.6	51.4	858
Do you subscribe to any environmental/scientific/animal magazines?	6.3	93.7	858
Are you a member of any type of conservation/environmental group?	4.4	95.6	858

Answers are represented as percentages.

(**Table 11**). Those participants who said they were aware were then asked to indicate which country(ies) are still involved in whaling. Countries indicated by participants included: Japan, China, Greenland and the Faroes, Norway, Iceland, Canada, Russia, the United States, South Africa, Australia, New Zealand, Vietnam, South Korea, Sri Lanka, Sweden, Finland, Thailand, Somalia, Peru, India, and Indonesia. Of those participants, 56% said Japan and 5% said they were aware of countries participating in whaling but did not know which countries. Approximately half of participants indicated they would boycott visiting a country that was involved in whaling with the other half saying they would not (**Table 12**). Only 13% of respondents indicated they have heard of the International Whaling Commission (**Table 13**). Finally, over 85% of respondents indicated that they were opposed or strongly opposed to the hunting of

TABLE 11 | Responses to the question "Were you aware that there are several countries that are still involved in whale hunting (whaling)?"

	Yes	No
Percentage (%) of respondents	47.44	52.56

(n = 858).

TABLE 12 | Responses to the question "Would you boycott visiting a country involved in whaling?"

	Yes	No
Percentage (%) of respondents	48.25	51.75

(n = 858).

TABLE 13 | Responses to the question "Have you heard of the International Whaling Commission (IWC)?"

	Yes	No
Percentage (%) of respondents	13.52	86.48

(n = 858).

whales (**Figure 10**). Of note, 23.91% of participants from India supported or strongly supported the hunting of whales as compared to only 8.84% of participants from United States (**Figure 11**).

A chi-square goodness-of-fit test was conducted to determine if the sample proportions of how supportive respondents were to the hunting of whales were all the same. The study found a statistically significant difference between the levels of supportiveness ($x^2 = 265.346$, $df = 2$, $p \leq 0.001$). That is to say, the public was significantly more likely to be opposed to whaling.

DISCUSSION

Study Limitations

This survey was conducted in an online format due to time constraints with a sample size of 858 MTurk users 18 years and older. Users of MTurk also tend to come predominantly from two counties: the USA and India, and this was the case in this study. However, efforts were taken to separate and contrast the datasets from these two countries, to investigate differences in attitudes.

Wright (2005) examined the advantages and disadvantages related to conducting online surveys by reviewing current features, issues, pricing, and limitations associated with products and services including online questionnaire features and services used to facilitate the online survey process. One advantage of conducting internet-based survey research is the ability to reach thousands of people regardless of distance (Bachmann and Elfrink, 1996); data can be collected while other tasks are being performed thus saving time (Llivea et al., 2002; Andrews et al., 2003; de Leeuw, 2012). Additional advantages include the possibility of the survey being more cost effective by eliminating the cost of paper, printing, data entry, and postage (if applicable) (Bachmann and Elfrink, 1996; Llivea et al., 2002; de Leeuw, 2012) and the ability of the survey to reach groups that only exist in cyberspace and do not require face-to-face meetings (Wright, 2005). However, there are concerns surrounding the design, implementation, and evaluation of an online survey. Non-response is problematic in any type of survey. Compared to other methods, including in person interviews and postal surveys, online surveys generally yield a lower response rate (de Leeuw, 2012). Additionally, excessive survey length, lack of interest, and poor survey design can hurt online survey response rates (Dillman et al., 2009). Internet coverage/accessibility of the general population can also be an issue, especially under-coverage of certain sub-groups including the elderly and the less educated (de Leeuw, 2012).

Demographics

Trends exist in who responds to surveys, and who does not, with regard to traditional survey methods (Smith, 2008). In general, younger people are more likely to participate than older people (Moore and Tarnai, 2002); more women are likely to participate than men (Curtin et al., 2000; Moore and Tarnai, 2002); and more educated and more affluent people are more likely to participate than less educated, less affluent people (Curtin et al., 2000). Online surveys are relatively new by comparison but are quickly growing in importance (Dillman et al., 1999).

Most of the participants in this survey were born in the 1981–1990 time frame, which is similar to that of a study by Howard and Parsons (2006) where 21–30 year olds made up 23% of participants. Additionally, in a study looking at potential gender bias in online survey response, Smith (2008) found a difference in the online survey response rates of female and male participants, having higher response rates for females than males. However, the current study found a higher response rate from males ($n = 523$) than females ($n = 335$). Notably, there were a larger number of males between 25 and 34 years of age than females.

Knowledge, Attitude, and Behavior

This study showed that marine mammal conservation issues were important to a majority of the participants. It is possible that participants of this online survey had an interest in conservation or cetaceans and searched for a survey on MTurk using key words such as conservation, cetacean, dolphin, or public attitude. A majority of participants felt that marine mammal conservation laws and policies were important despite a lack of awareness of the conservation status of whales and dolphins. These findings suggest that detailed knowledge of the conservation status of whales and dolphins is not a prerequisite of strong positive attitudes toward cetacean conservation issues. It is possible that strong support for marine mammal conservation issues could be due to the fact that marine mammals are popular among the general public and produce positive, aesthetic, humanistic views (Kellert, 1999).

Additionally, the mean attitude and behavior levels of males and females were found to be significantly different. Kellert and Berry (1980) found that females possess a greater concern for the harassment of animals and seem to value animals as objects of affection, leading to stronger conservation attitudes. The mean attitude level also differed between level of education of the participants with a significant difference between those participants indicating they had a Master's degree and those

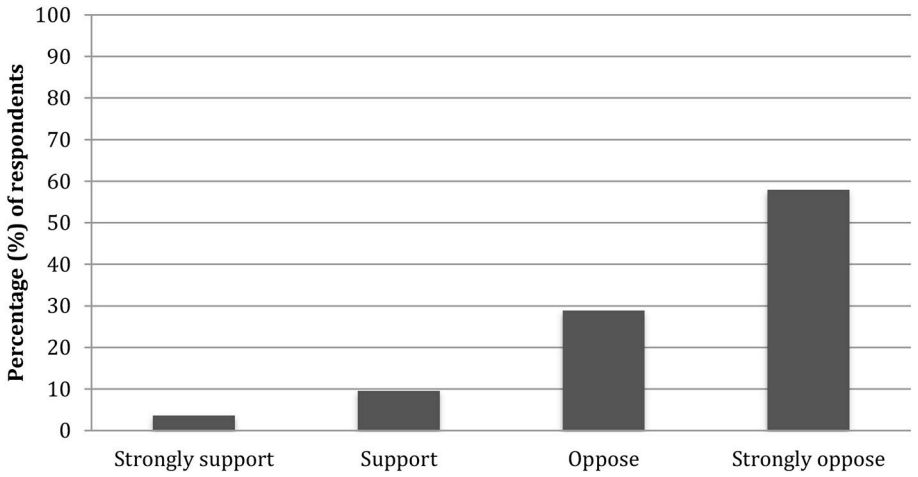


FIGURE 10 | Displays percentage of respondents whose level of support was either strongly supportive, supportive, opposed, or strongly opposed to the hunting of whales ($n = 858$).

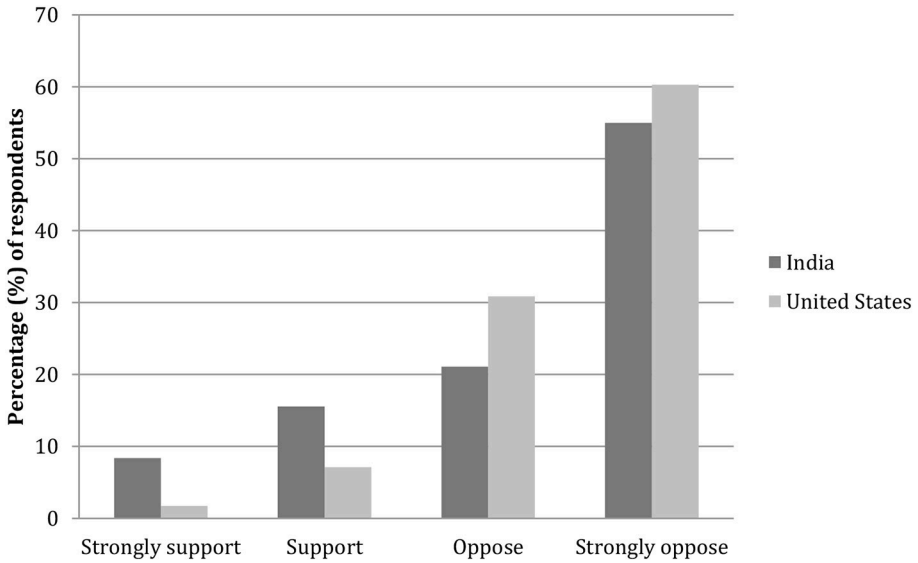


FIGURE 11 | Displays percentage of respondents from the United States and India whose level of support was either strongly supportive, supportive, opposed, or strongly opposed to the hunting of whales (US, $n = 577$; India, $n = 251$).

that had a high school diploma/ GED and some college, or no degree. These findings are supported by a study conducted by Kellert (1996) who found that education is most likely to change a person's attitude and perception of nature and biodiversity. Thompson and Mintzes (2002) and Kellert and Berry (1980) similarly found a direct relationship between education level and interest, concern, and awareness of environmental issues.

Consistent with findings by Barney et al. (2005), a positive correlation was observed in the current study between knowledge level, attitude level, and behavior level. Barney et al. (2005) found a person's knowledge of bottlenose dolphins increased with age and educational exposure, and were increasingly more likely to have an environmentally friendly, ecoscientific view of dolphins. Many researchers believe that knowledge and attitude are liked to each other and attitude is further connected to behavior (Flamm, 2006). When discussing the environment it can be assumed "if people become more knowledgeable about the environment and its associate, they will, in turn, become more aware of the environment and its problems and thus, be more motivated to act toward the environment in more responsible ways" (Fahlquist, 2009).

Conservation

Similar to the Howard and Parsons (2006) survey and the Scott and Parsons (2005) survey, the overall public concern reported in this survey, regarding dolphin and whale conservation, was

high. Almost 50% of participants in the current study thought that dolphin and whale conservation was very important; with another 46.63% indicating it was important or slightly important. In a survey investigating the awareness and attitudes of resident Arubans (in the Dutch Antilles, the Caribbean) and tourists toward marine mammals and their conservation, Luksenburg and Parsons (2013) found that 88.4% of participants felt marine conservation issues were important or very important, with only 2.5% indicating they were unimportant or very unimportant, and 9.0% indicated they had no opinion.

Given a list of whale species, 24% of current survey participants thought the blue whale was the most endangered, 22% thought the humpback whale was the most endangered, and 23% thought the killer whale was the most endangered. Of note, significantly more participants from the United States felt the humpback whale was the most endangered compared to participants from India (26 and 13.54% respectively) and significantly more participants from India felt the blue whale was the most endangered compared to participants from the United States (28.69 and 22.01% respectively). Only 14.33% of current survey participants felt the Hector's dolphin was the most important to conserve. In a preliminary study of American college students, Parsons et al. (2010) found that 39% of participants felt the humpback whale was the most threatened followed by the blue whale with 24.8%; only 4.8% indicated the North Pacific right whale was the most threatened. During the time the Parsons et al. (2010) survey was conducted, the North Pacific right whale was considered to have one of the worst conservation statuses and considered to be one of the most endangered whale species globally. In an additional study of college students by Sitar-Gonzales and Parsons (2012) on the perceived conservation status of polar bears and penguins, 69 and 53% of participants felt polar bears and penguins, respectively, were internationally classified as "endangered." At the time of the study, the polar bear was listed as "vulnerable" and five penguins were internationally classified as "endangered" and six as "vulnerable" (Sitar-Gonzales and Parsons, 2012).

The blue whale is highly recognizable to the general public. It is known as the largest animal on earth, possibly leading to its identification in this survey as the most endangered whale or dolphin species. Additionally, the blue whale often frequent the coast of India and is a popular species on whale-watching tours around Sri Lanka, possibly leading to the familiarity of the general public of India and leading to more Indian participants feeling the blue whale was the most endangered. The humpback whale, also very familiar to the general public, is often the main focus for the US whale-watching industry, and possibly attracts more media attention than any other large whale (Parsons et al., 2010). The killer whale is listed as "data deficient" by the IUCN but southern resident killer whales are listed as endangered by the US Endangered Species Act (ESA). It is possible that the highly publicized book *Death at SeaWorld* by David Kirby, or the much-viewed documentary *Blackfish* could have influenced the high percentage of participants in the current study that felt the killer whale was the most endangered cetacean species. Similar to the Parsons et al. (2010) study in which only a very small number of participants correctly identified the most endangered listed cetacean, the current study only had a small number of participants indicate that the Hector's dolphin was the most important to conserve. The Hector's dolphin is listed as "endangered" by the IUCN and a candidate species throughout its range by the ESA. It has one of the most restricted distributions of any cetacean and has suffered and is drastic decline over the past 30 years with levels of mortality being unsustainable (Reeves et al., 2013).

Participants in the current study were also asked which species listed was the most important to conserve. Almost 40% of respondents indicated the bottlenose dolphin was the most important to conserve, with more participants from the United States indicating this than participants from India. The bottlenose dolphin is listed as a species of "least concern" by the IUCN and is not listed under the ESA (although the Fiordland population in New Zealand is a candidate for ESA listing). Therefore, its identification as the whale and dolphin species most important to conserve was surprising. Similar to the humpback whale, the bottlenose dolphin is very recognizable to the general public, especially in the United States, being one of the most common cetacean species in captivity. Despite their high media profile, awareness of the bottlenose dolphin conservation status, in the sample population, was low, which is consistent with findings on polar bears and penguins from the Sitar-Gonzales and Parsons (2012) study. It was also notable that 16% of respondents felt the whale shark was the most important to conserve. Additionally, significantly more participants from India indicated the whale shark was the most important to conserve than participants from the United States. While the whale shark is listed as "vulnerable" by the IUCN, and is petitioned to be listed as either threatened or endangered by the ESA, it is a fish not a cetacean. The whale shark was heavily exploited during the 1990s off the Gujarat coast of India (Pravin, 2000) but as of 2001 it is protected under the Indian Wildlife (Protection) Act [Wildlife Trust of India (WTI, 2013)], thus possibly leading to the increased number of Indian participants who felt the whale shark was the most important to conserve. It is also possible that respondents did not accurately read the question, or rather disturbingly, they incorrectly viewed the whale shark as a cetacean. Similarly, Scott and Parsons (2004) found that general awareness of the occurrence and diversity of cetaceans in southwest Scotland was low. Participants were show photographs of the four most commonly occurring species; only 30.2% of participants identified one or more of the species correctly. The species most correctly identified in the study was the bottlenose dolphin (19%), followed by the harbor porpoise (17.5%), minke whale (10.7%), and lastly the common dolphin (7.1%) (Scott and Parsons, 2004).

On the list of species presented during this survey, having one of the worst conservation statuses, the Vaquita porpoise is listed as "critically endangered" by IUCN and "endangered" by ESA. Despite being the same species, the Vaquita porpoise received more votes than under an alternative common name of [Gulf of] California harbor porpoise. However, only 6.29% of respondents collectively felt it was the most important dolphin and whale species to conserve. The three "fake" species (the Lump-headed dolphin, the Majestic spotted dolphin, and the

Pygmy short-finned whale) received more votes collectively than the [Gulf of] California harbor porpoise, the Vaquita porpoise, the Northern right whale, and the Fin whale (17.37 and 16.02%, respectively). Of the three "fake" species, more participants thought the Majestic spotted dolphin was more important to conserve than the Lump-headed dolphin and the Pygmy short-finned whale (7.0, 6.17, and 4.2%, respectively). These results might be expected as a study by Karaffa et al. (2012) found that on average species names with negative connotations gathered less support for conservation (51%) whereas charismatic/positive sounding names prompted greater support for conservation (65%). Respondents were twice as likely to not conserve a negative sounding species (14%) than a positive sounding species (7%) (Karaffa et al., 2012).

Scott and Parsons (2005) investigated levels of public awareness of cetacean protection finding that 45.6% of participants felt cetaceans in Scotland were not sufficiently protected. Additionally, participants were asked to comment on the level of threat posed to cetaceans in Scottish waters by a variety of activities; oil spills (43.7%), reduction in available prey (41.8%), marine litter (32%), and sewage bacteria (31.1%) were indicated by most participants as posing the greatest threat (Scott and Parsons, 2005). Only 0.8% considered whale watching to be a serious threat (Scott and Parsons, 2005). Similarly, Howard and Parsons (2006) found 33% of participants felt cetacean's protection in Scotland was insufficient. Oil spills (68%), chemical pollution (65%) sewage pollution (63%), depletion of cetacean prey from over-fishing (54%), entanglement in fishing gear (51%), marine litter (44%), and global warming (43%) were considered by participants to be the greatest threats to cetacean populations, with only 3.1% indicating whale watching was a serious threat (Howard and Parsons, 2006). The current study found that 30% of people felt cetaceans were under protected with an additional 42% indicating cetaceans were only slightly protected.

Whaling

Scott and Parsons (2005) noted that 69.4% of participants stated they were aware of commercial and 'scientific' whaling operations being conducted. Almost all participants were against hunting whales (94.4%) with no participants indicating they strongly supported whaling. A large majority of this study's participants (86.83%) indicated they either strongly opposed or opposed hunting with 3.61% of participants strongly supporting whaling. Freeman and Kellert (1992) commissioned Gallup Organization to conduct an opinion poll on whaling in six countries including Australia, England, Germany, Japan, Norway, and the United States. They found that participants from the four non-whaling countries (Australia, England, Germany, and the United States) were highly opposed to whaling as compared to participants from the two whaling countries (Japan and Norway) where only a minority was opposed to whaling. The public of the whaling countries knew more about whaling, but knowledge about whale populations did not differ between the whaling and non-whaling countries. In 1999, Kellert conducted another study of American attitudes finding that 70% of Americans opposed whaling.

Only 48.25% of respondents indicated they would not visit a country involved in whaling, with slightly more Americans indicating they would not visit than Indians (49.05 and 48.61%, respectively). This is contrary to a study by Parsons and Rawles (2003) who found that 79% of whale-watchers in Tobermory, Isle of Mull, Scotland, would boycott a country that conducted hunts for cetaceans. Parsons and Draheim (2009) also found the 77.1% of tourists in the Dominican Republic would be less likely to visit a Caribbean country on vacation that supported the hunting or capture of whales and dolphins.

CONCLUSIONS

This study provides an initial indication of online survey participant views toward dolphins and whales, and their conservation.

The results of this study revealed high public concern regarding dolphin and whale conservation. Most participants felt marine mammal conservation laws and policies were important with a large portion of participants indicating that dolphin and whale species are only slightly protected or are under protected.

An over whelming majority of survey participants felt that whale and dolphin conservation was important; however, similar to previous studies, this study suggests a lack of awareness of the actual conservation status of several high-profile species including the bottlenose dolphin, humpback whale, and killer whale. Additionally, the results suggest there is a lack of public awareness of the conservation status of the "critically endangered" Vaquita porpoise, the "endangered" Northern right whale, and the "endangered" Fin whale. Furthermore, the Vaquita porpoise received more votes than by its alternative common name of [Gulf of] California harbor porpoise suggesting that an alternative name or the use of just one name might make social marketing campaigns more effective. Public concern for the three "fake species" exceeded that of actual species of conservation concern. This study noted a possible public misconception about the whale shark being a cetacean rather than a fish species. This study suggests that support for marine mammal conservation issues does not depend on detailed knowledge of the actual conservation status of cetacean species; however, the lack of connection between public awareness and the conservation status of whales and dolphins is concerning and certainly makes conservation efforts more difficult. Greater outreach to the public about the conservation status of whale and dolphin species is recommended.

Widespread opposition was expressed among most participants toward whaling.

This finding is consistent with the results of other studies, although some research has suggested support for the harvest of whale species in countries that currently participate in whaling. Opposition to whaling in the current study occurred among all demographic groups. It is possible that there is an increased awareness that living cetaceans are a valuable resource such as in countries that have whale-watching activities or increased media attention of marine mammals has increased public appeal

(Scott and Parsons, 2005). However, only half the participants indicated they were aware that several countries are still involved in whaling. Increased outreach efforts to the general public by groups dealing with whaling issues may be necessary.

The current study only surveyed a restricted group consisting of individuals over the age of 18 utilizing MTurk. Participation was not exclusive to individuals in the United States even though a majority of participants reported the US as their permanent place of residence. A larger sample size could increase reliability of the results and conducting an in-person survey could reduce inaccuracies in provided demographic and characteristic information. The current study results nevertheless suggest that there is a lack of awareness of the conservation status of certain whale and dolphin species, the existence of the IWC and countries that participate in whaling.

ETHICS STATEMENT

The study was reviewed and approved by the George Mason University Human Subjects Review Board.

AUTHOR CONTRIBUTIONS

WN conducted the survey and analysis and wrote the first draft of the manuscript. EP was WN's MS thesis advisor and devised the idea for the project, helped design the survey, provided methodological and analysis advice, and edited the manuscript and provided additional text.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2018.00153/full#supplementary-material

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Present and Potential Future Distributions of Asian Horseshoe Crabs Determine Areas for Conservation

Stine Vestbo[1]*, *Matthias Obst*[2], *Francisco J. Quevedo Fernandez*[3], *Itsara Intanai*[4] and *Peter Funch*[1]

[1] Genetics, Ecology, and Evolution, Department of Bioscience, Aarhus University, Aarhus, Denmark, [2] Department of Biological and Environmental Sciences, University of Gothenburg, Gothenburg, Sweden, [3] School of Computer Science and Informatics, Cardiff University, Cardiff, United Kingdom, [4] Department of Biology, Faculty of Science and Technology, Prince of Songkla University, Pattani, Thailand

Conservation of horseshoe crabs has recently received increasing attention as several populations are in decline. However, scarce information on their distributions in Southeast Asia is impairing conservation efforts. In this study, we sought to improve our understanding of the geographical range and distinct populations of the three Asian horseshoe crabs species in order to identify optimal conservation areas. We mapped the geographic range of *Carcinoscorpius rotundicauda, Tachypleus gigas,* and *T. tridentatus* using recent data from field work, literature, Global Biodiversity Information Facility (GBIF), and unpublished data from our scientific network. The data were correlated with 23 different environmental variables of potential ecological importance for horseshoe crabs using the openModeller webservices, including new tidal variables. Ecological niche models were generated using two algorithms, Maximum Entropy and support vector machine, for the three species under present conditions, and projected into a climate change scenario of 2050. The niches of the Asian horseshoe crabs were mostly determined by tidal regime, chlorophyll A concentrations, depth, distance to land, and sea surface temperature. According to our predictions, horseshoe crabs in Southeast Asia are not expected to experience any severe change in extent and distribution of suitable habitat in the future. In order to conserve Asian horseshoe crabs, we suggest establishing Marine Protected Areas at locations where distinct populations and several species occur, such as northern Vietnam, China, Borneo, and southern Japan.

Keywords: ***Carcinoscorpius*, climate change, ecological niche modeling, conservation, marine protected area, *Tachypleus*, Xiphosura**

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Natural Resources Defense Council,
United States

***Correspondence:**
Stine Vestbo
vestbo@bios.au.dk

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INTRODUCTION

Coastal and intertidal areas are currently under threat globally due to a range of anthropogenic activities, including infrastructure development and coastal protection, as well as effects of climate change, such as rising sea levels pushing coastal areas closer to anthropogenic structures, resulting in "coastal squeeze" (Defeo et al., 2009). Coastal areas are characterized by high productivity and species richness (Ketchum, 1972; Ray, 1991), and humans depend strongly on a variety of

species that are living and reproducing in the coastal zone, such as fishes (Allison et al., 2009; Bell et al., 2009), sea cucumbers (Purcell et al., 2013), and horseshoe crabs (Chatterji, 1994), thus the conservation of such species is of great importance.

During the past decade, horseshoe crab conservation has received increasing attention (Botton et al., 2015). Horseshoe crabs are marine chelicerates of the order Xiphosura, containing only four extant species. Three species occur in the coastal waters of India, Southeast Asia, China, and Japan [*Carcinoscorpius rotundicauda* (Latreille, 1802), *Tachypleus gigas* (Müller, 1785), and *T. tridentatus* (Leach, 1819)]. One species [*Limulus polyphemus* (Linnaeus, 1758)] occurs in the coastal waters of the eastern continental shelf of North America and in the Gulf of Mexico (Sekiguchi and Shuster, 2009). Populations of horseshoe crabs world-wide are currently in decline (e.g., Cartwright-Taylor et al., 2011; Kwan et al., 2016; Smith et al., 2017), and the IUCN Red List of Threatened Species now describes *L. polyphemus* as vulnerable (Smith et al., 2016). Although the Asian horseshoe crab species are thought to be in greater decline than *L. polyphemus*, these species are currently listed as data deficient (World Conservation Monitoring Centre, 1996a,b,c). While the geographic range of *L. polyphemus* is known (Sekiguchi, 1988; Anderson and Shuster, 2003; Sekiguchi and Shuster, 2009; Faurby et al., 2011), the precise distribution of the three Asian species remains less clear (World Conservation Monitoring Centre, 1996a,b,c), thus impairing conservation efforts.

The global decline in horseshoe crab populations is mainly due to anthropogenic activities. For *L. polyphemus* the decline has primarily been due to overharvesting and loss of breeding habitat (Smith et al., 2017), as this species breeds on beaches with slight slopes; a habitat commonly used for real estate construction and development (Nordstrom, 2004). Coastal armoring as a response to erosion is an additional factor reducing available breeding habitat (Jackson et al., 2015). Wild caught horseshoe crabs are used commercially in medicine, where substances from their hemolymph (*Carcinoscorpius, Limulus,* and *Tachypleus* Amoebocyte Lysate, CAL, LAL, and TAL) are used to test if drugs, blood products, and pharmaceutical devices are free from bacterial contamination (Levin and Bang, 1968; Rao and Bhagirathi, 1989; Levin et al., 2003). *L. polyphemus* is released after collection of LAL, but the two *Tachypleus* species are commonly used for chitin production after hemolymph collection, thus harvesting of these species results in 100% mortality (Gauvry, 2015). However, a synthetic, commercially available version of the CAL test has been developed (Ding et al., 1995; Lonza, 2016). The three Asian horseshoe crab species are fished for human consumption (Botton, 2001), and were previously imported to North America to be used as bait (Smith et al., 2016), although this import is now prohibited (IUCN, 2013). As for *L. polyphemus*, the expansion of urban infrastructure into coastal areas, as well as coastal armoring present additional anthropogenic habitat disturbances for the Asian horseshoe crabs, thus in some areas, such as Japan and Peninsular Malaysia, horseshoe crab breeding habitats are now almost completely lost due to coastal infrastructure development (Botton, 2001; Nelson et al., 2016). In response to the declining horseshoe crabs populations, Marine Protected Areas (MPAs) have been established, especially in the USA (Carl N. Shuster Jr. Horseshoe Crab Reserve, Delaware Bay) and Japan (Saikai National Park in Nagasaki). However, in most of Southeast Asia MPAs protecting critical horseshoe crab habitats are still scarce.

Climate change might have additional detrimental effects on the distribution and population sizes of horseshoe crabs. Rising sea levels could reduce suitable horseshoe crab breeding habitats, since the retreating shorelines would bring the horseshoe crabs closer to the human infrastructure near the coasts. This is the case for Delaware Bay, New Jersey, USA, where sea levels have risen at a high rate during the twentieth century and hence decreased horseshoe crab breeding habitat (Loveland and Botton, 2015). Climate change would also affect ecological factors that influence the developmental success of horseshoe crab eggs and larvae, such as salinity and temperature (Jegla and Costlow, 1982; Laughlin, 1983; Ehlinger and Tankersley, 2004; Zaleha et al., 2011), oxygen levels (Palumbi and Johnson, 1982; Funch et al., 2016), beach geochemistry, wave energy, and erosion (Botton et al., 1988; Penn and Brockmann, 1994; Jackson et al., 2008). One of the species, *C. rotundicauda*, is found in habitats characterized by mangroves, which are also vulnerable to the effects of climate change, primarily rising sea levels (Gilman et al., 2008).

Ecological niche modeling (ENM, also known as species distribution modeling) allows for predictions of suitable habitats for a given species by identifying the environmental factors driving the species' distribution, such as temperature, salinity, or depth (Peterson et al., 2011). Furthermore, ENM can be used to project the distribution of suitable habitats of a species into future scenarios (e.g., future climate change scenarios), allowing forecasts of possible species range shifts. For example, the method has been used to predict future distributions and range shifts for Mediterranean fishes (Albouy et al., 2013) as well as changes in suitable habitat for commercially important fish and invertebrates along the Atlantic coast of North America (Kleisner et al., 2017). ENM can be regarded as an important tool for conserving threatened species, and has proven useful for the assessment and planning of protected areas (Kremen et al., 2008; Stirling et al., 2016). Thus, the scattered information on Asian horseshoe crab distributions and their unknown conservation status would benefit from gathering recent occurrences as well as a modeling of suitable habitats in the region. The objectives of this study were to (1) identify important environmental factors that influence the distributions of the three Asian species of horseshoe crabs, (2) update the information on current species distributions, and (3) identify possible high priority regions for their conservation, where climate change has minimal effect and where species ranges overlap.

MATERIALS AND METHODS

Species Data and Study Area

The geographic study area included all coastal waters between 10.0° S to 35.0° N and 77.0° W to 135.0° E, thus including the continental shelves of the Bay of Bengal, India and Indonesia and between Indonesia and Japan. This area, the geographical mask, was chosen based on the existing knowledge

of horseshoe crab distribution limits in Asia (Sekiguchi, 1988), and used for geographical filtering of occurrence records, collecting background points, and building and projecting the models. We assembled occurrence records (presence-only) for all three species in this region from our own observations, collaborators, scientific networks as well as through publishing a scratchpad site at http://horseshoecrabs.myspecies.info/ (**Table 1**, **Supplementary Table 1**). For many species, numerous distribution records exist in the literature, and we manually geo-referenced additional occurrence data from these sources (**Supplementary Table 2**). Subsequently, we added data from the Global Biodiversity Information Facility (http://gbif.org) in September, 2015 (**Supplementary Table 3**). A taxonomic data refinement workflow (Mathew et al., 2014) was used to check the synonyms, download, visualize, filter, and integrate occurrence records for all species. We inspected all records and excluded those of dubious identity (e.g., on land, in deep water, or without clear reference of origin), as well as those older than 1995 and those outside the geographical mask. All new data (i.e., excl. GBIF records) were submitted to the Ocean Biogeographic Information System (http://www.iobis.org/) and are available under the link https://doi.org/10.14284/293.

Environmental Data

Based on available information on general ecology of horseshoe crabs (Bonaventura et al., 1982; Sekiguchi, 1988; Shuster et al., 2003), we included 23 environmental variables in the analysis, relating to sea surface temperature, chlorophyll A concentration, salinity, oxygen and nutrient concentrations, distance to land, water depth, pH, and tidal regime (**Table 2**). Chlorophyll A variables were included since chlorophyll levels acts as a proxy for phytoplankton primary production, which in turn reflects food availability for filter-feeding bivalves, which is one of the most important food items for horseshoe crabs (Chatterji, 1994). We used 13 marine layers from Bio-Oracle (http://www.bio-oracle.ugent.be/) with a resolution of five arc-minutes (**Table 2**; Tyberghein et al., 2012). These data layers are generated from monthly satellite data (Aqua-MODIS and SeaWiFS; https://oceancolor.gsfc.nasa.gov) as well as *in situ* measured oceanographic data from the World Ocean Database, 2009 (Boyer et al., 2009), and overlap with the species occurrence data in their temporal origin. In addition, we used five marine layers for both present and 2050 climate scenarios from AquaMaps (http://www.aquamaps.org/download/main.php) with a resolution of 30 arc-minutes (**Table 2**; Kaschner et al., 2008). Present-day (i.e., year 2010) datasets from AquaMaps were built from long-term averages of temporally varying environmental variables (Ready et al., 2010), while future data sets for 2050 were derived from the IPSL-CM4 A2 climate change scenario (IPCC, 2007).

We additionally generated five present-day (i.e., year 2010) global tidal variables with the Finite Element Solution oceanographic model (FES2012), provided by Noveltis, Legos and CLS Space Oceanography Division and distributed by AVISO+ (http://www.aviso.altimetry.fr/) (**Table 2**, more detailed information on how the global tidal variables were created can be found in **Supplementary Data Sheet 1**). FES2012 is a fully revised version of the original global hydrodynamic tide solution

TABLE 1 | Occurrence data used in the analysis. Environmentally unique points (EUPs) were calculated from the compiled observations (records) using the respective geographical mask with a resolution of 300 arc seconds and 23 environmental variables.

Species	Records (EUPs)	Source references (number of records)
Carcinoscorpius rotundicauda	123 (67)	Adibah et al. (2015) (5), Behera et al. (2015) (1), Cartwright-Taylor et al. (2009) (1), Cartwright-Taylor et al. (2011) (7), Chatterji (1999) (2), Chatterji and Parulekar (1992) (1), Chiu and Morton (2003) (2), Dao et al. (2009) (1), Faizul et al. (2015) (1), Faurby et al. (2011) (3), Fusetani et al. (1982) (1), GBIF (11), Hong (2004) (1), Jeffries et al. (1989) (1), Key et al. (1996) (1), Kungsuwan et al. (1987) (3), Lee and Morton (2005) (1), Ngy et al. (2007) (1), Raman et al. (2014) (2), Robert et al. (2014) (2), scientific network (50), Sekiguchi (1988) (15), Srijaya et al. (2010) (2), Tanu and Noguchi (1999) (1), www.wildsingapore.com (3), Yap et al. (2011) (3), Zhou and Morton (2004) (1)
Tachypleus gigas	125 (88)	Behera et al. (2015) (1), Cartwright-Taylor et al. (2011) (1), Chatterji (1999) (1), Chatterji et al. (1992) (1), Chatterji et al. (2004) (1), Faizul et al. (2015) (1), Faurby et al. (2011) (2), GBIF (24), Ismail and Sarijan (2011) (2), John et al. (2011) (2), Kamaruzzaman et al. (2011) (1), Key et al. (2000) (3), Kungsuwan et al. (1987) (3), Liew et al. (2015) (5), Patil and Anil (2000) (1), Raman et al. (2014) (2), Robert et al. (2014) (1), Rozihan and Ismail (2011) (4), Rozihan and Ismail (2012) (3), Sahu and Dey (2013) (1), scientific network (51), Sekiguchi (1988) (8), Shakibazadeh et al. (2013) (1), Tan et al. (2011) (1), Vijayakumar et al. (2000) (1), Zaleha et al. (2011) (1), Zaleha et al. (2012) (2)
Tachypleus tridentatus	81 (55)	Almendral and Schoppe (2005) (1), Botton et al. (1996) (2), Cai et al. (2015) (1), Chatterji and Pati (2014) (1), Chiu and Morton (2003) (5), Chiu and Morton (2004) (1), GBIF (6), Hsieh and Chen (2009) (4), Hu et al. (2009) (6), Itow et al. (1998) (4), Kannan et al. (1995) (2), Kwan et al. (2015) (1), Lee and Morton (2005) (1), Liao et al. (2012) (1), Nishida et al. (2015) (1), Robert et al. (2014) (2), scientific network (23), Shigenaga et al. (1990) (1), Weng et al. (2012) (5), Xu et al. (2011) (3), Yang and Ko (2015) (1), Yang et al. (2009a) (3), Yang et al. (2009b) (1), Yang et al. (2007) (3), Zhou and Morton (2004) (2)
Sum	329 (210)	

All original occurrence records are accessible through http://www.iobis.org/.

TABLE 2 | Environmental variables investigated for ecological niche modeling of the three Asian horseshoe crabs.

Source (Resolution)	Variable (Abbreviation)	Unit	CR	TG	TT
Bio-Oracle (5 arc-minutes)	Mean dissolved oxygen (Oxy)	ml/l			
	Mean nitrate $[NO_3]/[NO_3+NO_2]$ (NO3)	µmol/l			
	Mean phosphate (PO4)	µmol/l			
	Silicate	mol/m^3			
	pH	–			
	Mean sea surface salinity (SSS)	PSU			
	Maximum sea surface temperature (Max SST)	°C			
	Minimum sea surface temperature (Min SST)	°C			
	Sea surface temperature range (Range SST)	°C			*
	Mean calcite concentration (CaCO3)	mol/m^3			
	Maximum chlorophyll A concentration (Max ChlA)	mg/m^3	*	*	*
	Minimum chlorophyll A concentration (Min ChlA)	mg/m^3			
	Range of chlorophyll A concentration (Range ChlA)	mg/m^3	*	*	*
AquaMaps (30 arc-minutes)	Mean sea surface salinity (SSS)	PSU			
	Mean sea surface temperature (Mean SST)	°C	**	**	**
	Mean chlorophyll A concentration (Mean ChlA)	mg/m^3	**	**	**
	Distance to land (DL)	km	**	**	**
	Mean depth (Depth)	m			
FES2012 (3.75 arc-minutes)	**Annual average cycle amplitude** (AverAmpl)	cm	*	*	*
	Maximum annual cycle amplitude (MaxAmpl)	cm	*	*	*
	Annual standard deviation of cycle amplitude (CycleDev)	cm			
	Annual average duration of tidal cycles (CycleDur)	hours			
	Annual number of cycles (CycleNumber)	-			

Informative variables are shown in bold. Variables used in the modeling of present distribution (*) and potential range shifts (**) are indicated for each species under the columns CR (Carcinoscorpius rotundicauda)*, TG* (Tachypleus gigas)*, and TT* (T. tridentatus)*. Variable abbreviations used throughout the article are given in parentheses.*

model (Lyard et al., 2006) with improvements regarding longer altimeter time series, more accurate ocean bathymetry, major non-linear tides, and more accurate tidal currents. It includes overall 32 tidal constituents distributed on 1/16° grids (amplitude and phase), corresponding to 3.75 arc-minutes (Carrère et al., 2012). The tidal variable layers can be accessed through a webservice (https://www.biodiversitycatalogue.org/services/37), and is hence available for future research on distribution modeling of marine intertidal organisms. In addition, we deposited the raster files together with the algorithm for calling the FES program and creating the ASCII grids at the Swedish Environmental and Climate Data Repository (www.ecds.se) under the identifier 87196c86-8e36-4908-8f16-424e61aa313c.

Analysis of Environmental Variables

Environmental values were obtained using version 2 of the BioClim workflow (http://purl.ox.ac.uk/workflow/myexp-3725.2) available at the BioVeL portal, https://portal.biovel.eu (Hardisty et al., 2016). The workflow was used to retrieve environmentally unique points (EUPs) from the species occurrence files for a set of 23 environmental data sets (**Table 2**). Since all points within a raster cell share the same values for the environmental variables, EUPs are obtained by filtrating the occurrence points, so that each raster cell contains no more than a single point (Nix, 1986). The workflow returned environmental values from the specified layers that matched with the species occurrence records, and it was executed in batch mode (called data sweep function) to repeat all calculations with the same parameter settings for all species.

We analyzed differences in environmental tolerances between all three horseshoe crab species by applying a Welch's test for unequal variances to the retrieved environmental values (**Supplementary Table 4**). This test was chosen because the data did not conform to parametric test assumptions. In addition, pairwise comparisons of environmental tolerances between species were performed using the Games-Howell *Post-Hoc* test (**Supplementary Table 5**). Both statistical tests were performed using SPSS version 22 (IBM Corporation, 2013). We subsequently performed a correlation analysis (data not shown) as well as a principal component analysis (PCA) on the environmental values using the R statistical environment 3.0.2 (R Core Team, 2013). The analyses were performed in order to identify ecologically important variables that explain the variation in the data set, and which can be used as predictor variables (Metzger et al., 2005; Porfirio et al., 2014). The PCA was estimated as a 23-dimensional hypervolume and used to compare

the environmental space occupied by the three species. We used all non-correlated variables that were represented (with >5%) in the first two components of the PCA to build the niche models, except for Mean SST, which was included in the future projection (**Supplementary Tables 6–8**).

Ecological Niche Modeling

We used version 20 of the ecological niche modeling (ENM) workflow (http://purl.ox.ac.uk/workflow/myexp-3355.20) in batch mode (called data sweep) to describe and compare the geographical space occupied by the three species and to estimate the distribution of suitable habitat with favorable biotic, environmental, and geographical conditions. Demographic or dispersal properties that may also be used in species distribution modeling were not considered (Peterson et al., 2011; Reiss et al., 2014). For more information on the ENM workflows see Holl et al. (2013), Leidenberger et al. (2015a,b), and De Giovanni et al. (2015). We executed parallel analyses with two ENM algorithms by means of the openModeller webservice suite (de Souza Muñoz et al., 2011). These are i) Maximum Entropy v. 1.0 (Phillips et al., 2006; Phillips and Dudík, 2008) and ii) support vector machine v. 0.5 (Schölkopf et al., 2001). These two methods were chosen because Maximum Entropy (Maxent) is a presence-background algorithm that has been shown to perform with high predictive accuracy even if there are moderate errors in the geo-referencing (Elith et al., 2006; Graham et al., 2008), and support vector machine (SVM) is a machine-learning algorithm that is not analyzing characteristics of statistical distributions and hence does not require independent observation data, and thereby overcomes potential problems with spatial autocorrelation in the species occurrence data (Drake et al., 2006).

Models were created using each species' maximum distribution range within the mask and a set of predictor variables identified in the PCA. Variables selected for present day projections (high resolution variables) and future trend analysis (low resolution variables) are listed in **Table 2**. Models were created based on EUPs (**Table 1**) with the following specifications. Maxent models were set to run with 10.000 background points (including input points) drawn from the mask. Feature selection was automated, allowing the algorithm to combine feature types when fitting a model, and perform 500 iterations. Tolerance for detecting model convergence was set to 0.00001, while sample threshold was set to 80 (product), 10 (quadratic), and 15 (hinge). SVM models were set to execute the C-SVC algorithm with radial basis kernels, gamma values 1/k (where k is the number of layers), and a cost value of 1. All models were set to produce a probabilistic, instead of binary output. Predictive models may encounter some problems with extrapolations (Zurell et al., 2012), i.e., when the environmental hyperspace of the scenario is not represented by true observations. These were minimized by using only few essential variables in the future predictions, thereby avoiding complexity and interaction between too many factors, as well as through using a consensus approach (**Table 2**; Pearson et al., 2006).

For each species, we ran both algorithms across two sets of environmental layers (specified above and in **Table 2**); one with high resolution (using Bio-Oracle and FES2012 layers) and one with low resolution (using AquaMaps layers). The high-resolution models were used for present day projections of suitable habitat, while the low-resolution models were used to deduct climate driven changes in suitable habitat. Models were tested using 10-fold cross-validation based on the area under the curve (AUC) value and omission error rate (false negative rate), and subsequently projected using present and (in case of AquaMaps layers) 2050 climate scenarios (**Table 3**). The results of the ENMs were visualized as maps showing the distribution of suitable habitat of a species in a particular geographic region at a particular time (de Souza Muñoz et al., 2011). The sweep function allowed automated batch processing of all species for a given algorithm and parameter set. Overall, we executed 12 niche models, i.e., two algorithms, three species, and two environmental datasets (**Table 3**).

GIS Analysis

Raster maps created by the niche modeling algorithms were processed using the qGIS software package v. 2.6 Brighton (Quantum GIS Development Team, 2014). Maps of individual models are available in the **Supplementary Material** (**Supplementary Images 1–6**). For the consensus analysis presented in the maps, we transformed each individual model into a binary output using the lowest presence threshold (LPT) in order to include any signal of suitable habitat. Thereafter the maps of both models were merged into a single consensus model highlighting only areas where both algorithms agreed on the prediction of suitable habitat. Likewise, heat maps showing the predicted changes between 2050 and present-day distributions were created as consensus maps identifying only areas where both algorithms agreed on predicted loss or gain of suitable habitat. Finally, we compared high-resolution models (used for present day projections) with low-resolution models (for future change projections). We found that both model settings predict suitable habitat for similar biogeographic regions with few deviations mentioned in the discussion below.

RESULTS

Environmental Factors Defining Horseshoe Crab Distributions

The PCA showed that *C. rotundicauda* and *T. gigas* largely share the same ecological niche, and that the niche of *T. tridentatus* to some extent overlaps with the niches of the other two species (**Figure 1**). Further, the PCA showed that the distributions of the three species are mainly driven by chlorophyll concentration, tidal regime, temperature, depth and distance to land (**Figure 1**, **Table 2**). The combination of the environmental tolerances represents the specific fundamental ecological niche for each of the three species. We found no difference between the three Asian species in any of the tidal variables (**Figures 2A–C**). All species experienced varied annual cycle deviation as well as cycle durations of varied lengths. The mean minimum sea surface temperature found for *T. tridentatus* was 17.0°C and hence 10 degrees lower than those for *T. gigas* and *C. rotundicauda* (**Figure 2D**). *T. tridentatus* experienced a larger mean temperature range of 12.7°C compared to *T. gigas* and

TABLE 3 | Results of model tests for all models of current distribution (based on Bio-Oracle and tidal variables) and future trend analysis (based on AquaMaps variables).

Model	Algorithm	Criterion	Species		
			CR	TG	TT
Current distribution (BioOracle variables)	SVM	AUC	0.977 (0.01)	0.976 (0.01)	0.948 (0.04)
		OE	1.429 (4.52)	1.250 (3.95)	1.667 (5.27)
	MAXENT	AUC	0.983 (0.02)	0.985 (0.02)	0.935 (0.08)
		OE	2.857 (6.02)	1.111 (3.51)	2.000 (6.32)
Future trend analysis (AquaMaps variables)	SVM	AUC	0.959 (0.01)	0.967 (0.01)	0.962 (0.03)
		OE	1.667 (5.27)	1.250 (3.95)	3.333 (10.54)
	MAXENT	AUC	0.974 (0.03)	0.974 (0.02)	0.952 (0.07)
		OE	6.190 (8.03)	1.111 (3.51)	2.000 (6.33)

Performance was assessed for two model algorithms (SVM, Maxent) and all three species separately (CR, C. rotundicauda*; TG,* T. gigas*; TT,* T. tridentatus*), with area under the curve (AUC) values and omission error (OE) in %. Both criteria were measured as mean values obtained from 10-fold cross validation with standard deviation in parentheses.*

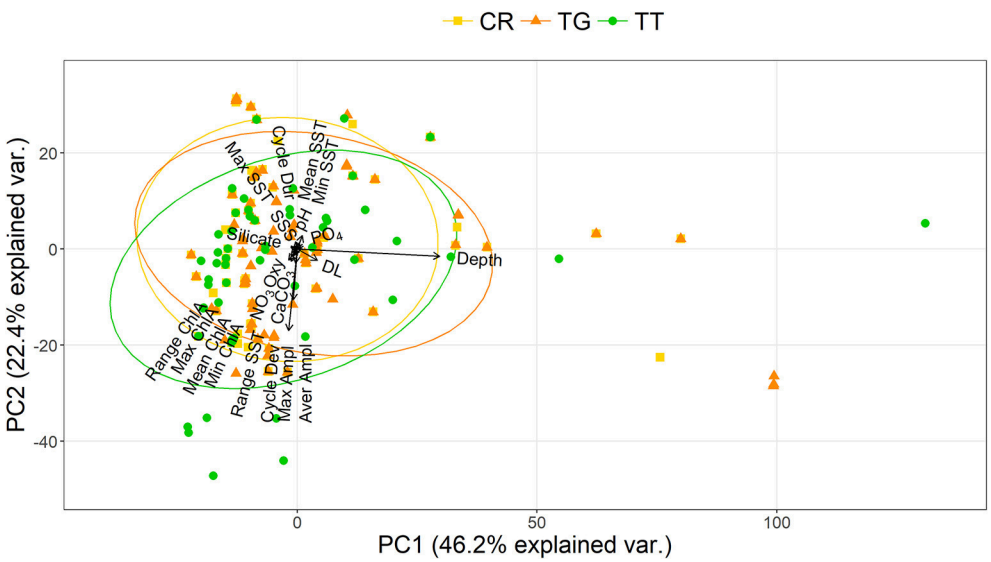


FIGURE 1 | Niche overlap analysis of the three Asian horseshoe crab species showing the principal component analysis plots of the environmental "hypervolume" for 23 predictor variables [CR, *C. rotundicauda* (■); TG, *T. gigas* (▲); TT, *T. tridentatus* (●)]. Ellipses represent 68% of the hypervolume for each species; points represent presence of each species at environmentally unique locations. The positions of variable names have been adjusted to prevent overlapping.

C. rotundicauda, which experienced a mean range of 3.9 and 4.0°C, respectively (**Figure 2E**). The water oxygen levels followed the same pattern as sea surface temperatures, with a higher mean for *T. tridentatus* (5.0 mL/L) compared to *T. gigas* and *C. rotundicauda* (4.4 and 4.5 mL/L, respectively, **Figure 2F**). Neither the maximum chlorophyll A concentration nor the range of chlorophyll A concentration differed between the species (**Figures 2G,H**). We found a mean salinity of 33.5 PSU for *T. tridentatus*, which was slightly higher compared to *T. gigas* (31.8 PSU) and *C. rotundicauda*. (32.3 PSU, **Figure 2I**), although the narrow salinity range for *C. rotundicauda* was likely an artifact caused by inaccessible data from inland waters (see Discussion). The mean nitrate level was higher for *T. tridentatus* (1.9 μmol/L) than the other two species (0.6 μmol/L for both) (**Figure 2J**). Mean phosphate differed between all three species, with *T. gigas* experiencing the highest (0.28 μmol/L) and *T. tridentatus* the lowest (0.20 μmol/L) mean level (**Figure 2K**). *T. tridentatus* differed from the other two species in occupying a niche with higher mean pH value (**Figure 2L**).

Species Distributions

The sampled distribution of *C. rotundicauda* includes the coasts of eastern India and Bangladesh, the coasts of Malaysia, Thailand, Cambodia, and southern Vietnam (**Figure 3**), but excludes the eastern coast of Vietnam, however, the distribution continues along the coast of northern Vietnam and southern China. *C. rotundicauda* is also present along the coasts of the Indonesian islands Sumatra and Java, as well as Borneo. The distribution of *T. gigas* is similar to that of *C. rotundicauda*, except that *T. gigas* seems to be absent from the coasts of Bangladesh, Cambodia, northern Vietnam, and China (**Figure 4**). *T. tridentatus* is distributed along the coasts of southern Japan, China, Taiwan, and northern Vietnam, in addition to the Philippines, and the islands of Borneo and Java (**Figure 5**).

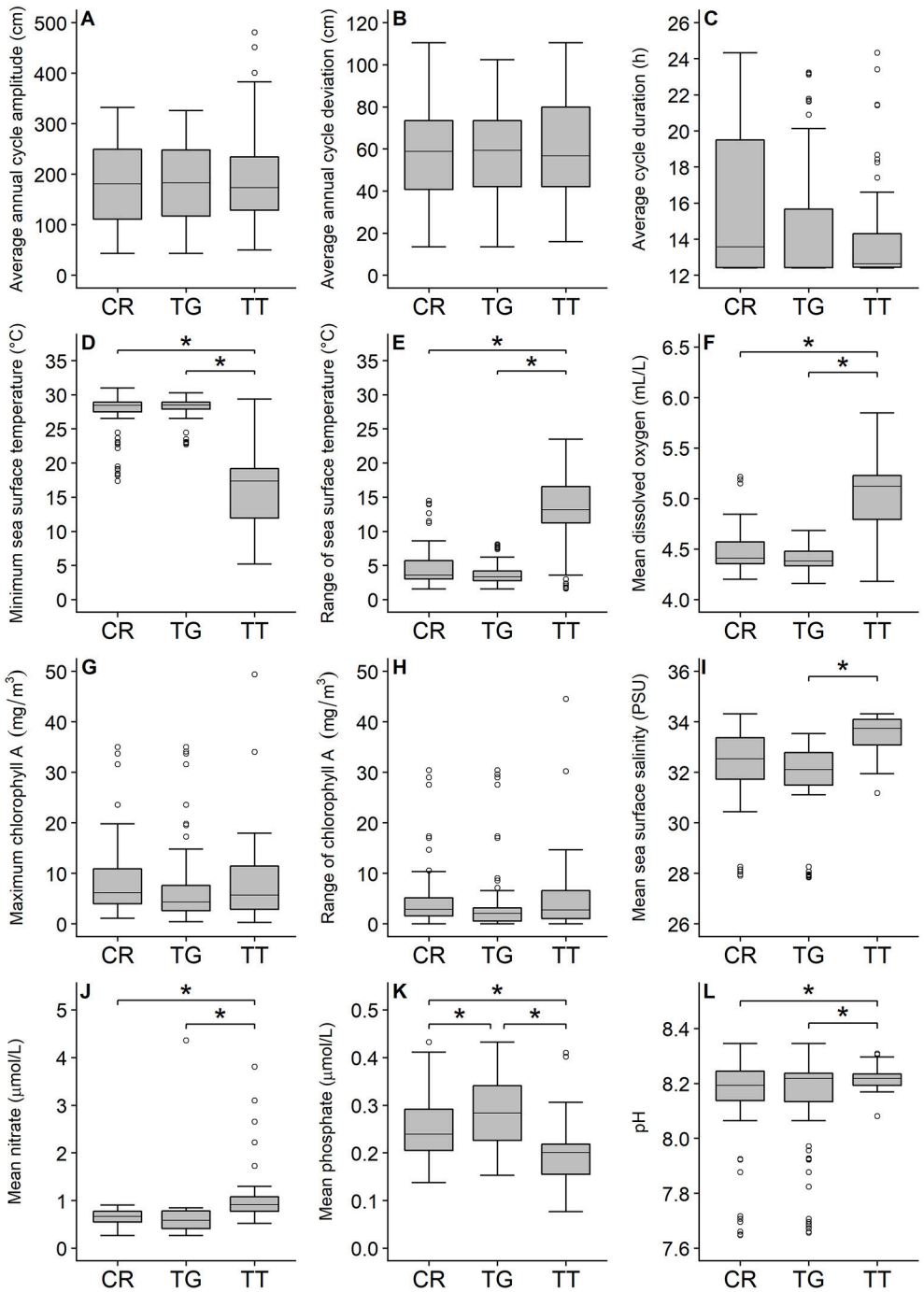


FIGURE 2 | Comparison of the three horseshoe crab species (CR, *C. rotundicauda*; TG, *T. gigas*; TT, *T. tridentatus*) showing the variation of 12 key environmental variables. Variable and unit are shown on the y-axis. Gray boxes indicate 50% of the sample points and are limited by the 1st **(Bottom)** and 3rd quartiles **(Top)**. Black, horizontal lines within the gray boxes display the medians. Comparisons between *C. rotundicauda* and the two *Tachypleus* species for salinity were excluded because of absence of environmental data for inland waters. Asterisks show significant (<0.05) differences between species.

Distribution of Suitable Habitat

Both models, SVM and Maxent, demonstrated high predictive power, as seen by the high AUC values (all values >0.9) (**Table 3**). The Maxent model performed marginally better than SVM for all species except *T. tridentatus*. Omission errors were very low for both models (<3.5%), except for Maxent's future prediction for *C. rotundicauda* (6.2%).

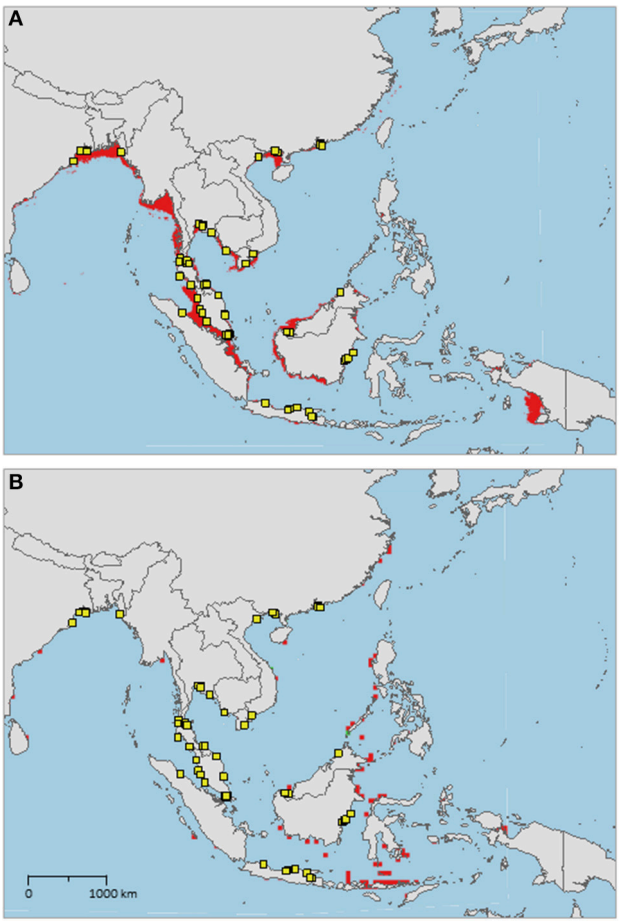


FIGURE 3 | Distribution maps for *Carcinoscorpius rotundicauda* with environmentally unique points, indicating the sampled distribution range for the species. Upper map **(A)** shows present-day projections of suitable habitat in red based on consensus of Maxent and SVM algorithms. Lower map **(B)** shows changes between the present-day and 2050 projections of suitable habitat based of consensus of Maxent and SVM algorithms, with green cells indicating loss of suitable habitat and red cells indicating gain of suitable habitat. Support values for the individual models are given in **Table 3**.

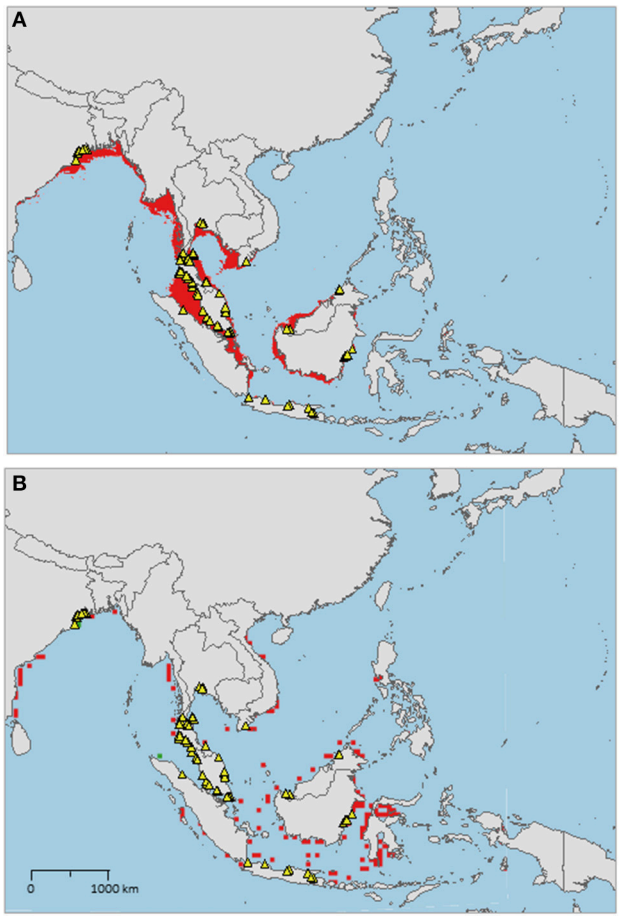


FIGURE 4 | Distribution maps for *Tachypleus gigas* with environmentally unique points, indicating the sampled distribution range for the species. Upper map **(A)** shows present-day projections of suitable habitat in red based on consensus of Maxent and SVM algorithms. Lower map **(B)** show changes between the present-day and 2050 projections of suitable habitat based of consensus of Maxent and SVM algorithms, with green cells indicating loss of suitable habitat and red cells indicating gain of suitable habitat. Support values for the individual models are given in **Table 3**.

Suitable habitats for *C. rotundicauda* are distributed along the coastline of mainland Southeast Asia, from the Bay of Bengal, including Sri Lanka, to southern China, except from the eastern coasts of Peninsular Malaysia and Vietnam (**Figure 3A**). Suitable habitats are also present in the Malacca Strait between Sumatra and Peninsular Malaysia, Southeast Sumatra, southern coast of Java, the coast of Kalimantan (Indonesian Borneo), and the coast of Sarawak (Malaysian Borneo). Additionally, small areas of suitable habitat are found in Sabah (Malaysian Borneo), West Papua, and in Manila Bay, Philippines. The distribution of suitable habitat differs from the sampled distribution, as it includes the coasts of Myanmar, southern Borneo, eastern Sumatra, and southern Java.

The distribution of suitable habitat for *T. gigas* is similar to that of *C. rotundicauda*, however, for *T. gigas* it does not go further east along the Southeast Asian mainland than to southeast Vietnam (**Figure 4A**), nor does it include the south coast of Java. Compared to the current known distribution, the distribution of suitable habitat includes the coast of Myanmar, Bangladesh, east coast of Sumatra, and southern Borneo.

The distribution of suitable habitat of *T. tridentatus* spans from northern Vietnam, along the coast of China, and into the Sea of Japan (**Figure 5A**). It also includes the coast of northwest Taiwan and Japan, including the Seto Inland Sea. Small areas of suitable habitat are present along the coasts of Southwest Myanmar, East Sumatra, South Java, Borneo, South Vietnam, Bay of Bangkok, and Manila Bay.

Changes in the Distribution of Suitable Habitat Under Climate Change Scenarios

We included three environmental variables, mean sea surface temperature, mean chlorophyll A concentration, and distance

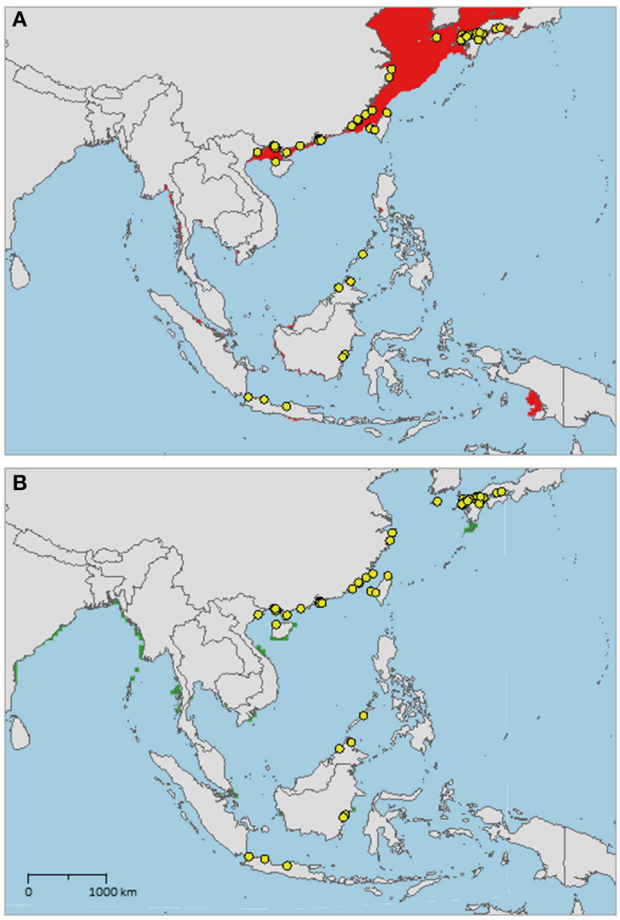


FIGURE 5 | Distribution maps for *Tachypleus tridentatus* with environmentally unique points, indicating the sampled distribution range for the species. Upper map **(A)** shows present-day projections of suitable habitat in red based on consensus of Maxent and SVM algorithms. Lower map **(B)** show changes between the present-day and 2050 projections of suitable habitat based of consensus of Maxent and SVM algorithms, with green cells indicating loss of suitable habitat and red cells indicating gain of suitable habitat. Support values for the individual models are given in **Table 3**.

to land, for the modeling of the future distributions of suitable habitat (**Table 2**). Our models predicted that the habitat suitability for *C. rotundicauda* will increase in several coastal areas throughout Southeast Asia. More suitable habitats will arise in more eastern regions; the Philippines and along the coasts of several Indonesian islands: Sulawesi, Lombok, Sumbawa, and Flores (**Figure 3B**). Similarly, for *T. gigas* several areas along the coasts of Southeast Asia will become more suitable in the future (**Figure 4B**). The Bay of Bengal and North Sumatra will be less suitable. *T. tridentatus* will lose suitable habitat in the future, however, none of the lost areas are overlapping with the species' current occurrences (**Figure 5B**). In summary, none of the Asian horseshoe crabs seem to undergo a major range shift driven by prospective changes of the environmental factors tested here.

DISCUSSION

Carcinoscorpius rotundicauda

We found that *C. rotundicauda* occurs along the coasts of Bangladesh, Cambodia, Vietnam, East Thai-Malay Peninsula, and China, in addition to the countries and coasts already surveyed by Sekiguchi (1988) (India, West Thai-Malay Peninsula, Singapore, and Indonesia). However, contrary to Sekiguchi (1988) we have not found evidence for its presence at Palawan, Philippines (**Figure 3**). According to the occurrence record, *C. rotundicauda* is absent along the coast of Myanmar, even though our models indicate suitable habitat in this area. Indeed, there is evidence that horseshoe crabs are present in this region, however, the available information points to *L. polyphemus*, and is likely a misidentification (Thapanand-Chaidee et al., 2010). We thus conclude that *C. rotundicauda* probably is present along the coast of Myanmar, and that missing records of horseshoe crabs are due to lack of studies in the area. The sampled distribution of *C. rotundicauda* shows a gap along the east coast of Vietnam, extending approximately 1,500 km in the high-resolution projections (based on Bio-Oracle variables, **Figure 3**) and approximately 400 km in the low-resolution projections (based on AquaMaps variables). The absence of the species in this region might be explained by the narrow continental shelf present along the coastline, which provides less horseshoe crab habitat, as well as water characterized by a high degree of mixing with deep water (Rojana-anawat et al., 2001). The distribution gap could also be related to food availability, as the east coast of central Vietnam has a lower macrozoobenthos biomass compared to the southern and northern coasts (Nguyen and Dao, 1995). However, the lack of mangroves along the east coast of central Vietnam may be the most plausible explanation for the distribution gap (Hong and San, 1993). *C. rotundicauda* is commonly known as the mangrove horseshoe crab, as it is often found in habitats characterized by mangroves and mudflats in contrast to the two *Tachypleus* species, which can be found at open sandy beaches (e.g., Cartwright-Taylor et al., 2011; Robert et al., 2014; Jawahir et al., 2017), thus a scarcity of mangroves would have a negative effect on the presence of *C. rotundicauda*.

When interpreting our model results, it is important to mention that species distribution models do have caveats, which may lead to false or uncertain projections of actual species distribution ranges (Jarnevich et al., 2015). We have tried to mitigate such caveats and uncertainties through an unbiased sampling of the biological data from independent sources, a careful selection of predictor variables, and through modeling with several environmental data sets and analytical algorithms. Nevertheless, the indication of a gap in the distribution of *C. rotundicauda* along the Vietnamese east coast should still be treated as a hypothesis that needs further validation with additional sampling and modeling.

The future distribution of suitable habitat of *C. rotundicauda* includes several coastal areas of Indonesian islands east of the sampled distribution. Since horseshoe crabs reside on the continental shelves (Sekiguchi and Shuster, 2009), it has been argued that the distributions of the Asian horseshoe crabs are delimited by the eastern edge of the continental Sunda Shelf

(roughly corresponding to Wallace Line). Thus, even though areas of future suitable habitat are present east of the Sunda Shelf, we would not expect *C. rotundicauda* to be able to reside here, unless humans have mediated their dispersal. The models used in this study find an approximation of the fundamental niche of the horseshoe crabs, not the realized niche, thus geographical barriers are not necessarily detected.

We found that *C. rotundicauda*, as well as the other two Asian horseshoe crab species, inhabit coastal areas with varied tidal regimes. This is in contrast to *L. polyphemus*, as this species can be found in areas with mixed semi-diurnal tides (i.e., tidal cycles of approximately 12 h) (Rudloe, 1985; Barlow et al., 1986). The narrow range of salinity found for *C. rotundicauda* is most likely an artifact, as we know that the three Asian species occur in river estuaries, and *C. rotundicauda* has been found at salinities as low as 10 PSU (Chatterji, 1999; Mishra, 2009; Chen et al., 2015). Unfortunately, we could not include occurrences from inland rivers in Southeast Asia in our models, due to lack of environmental data from these areas. The narrow temperature range of *C. rotundicauda* reflects its tropical distribution, and can be related to its preference for temperatures between 20 and 30°C (Srijaya et al., 2014), which seems to be necessary for regular ecdysis (Lee and Morton, 2005). We find that *C. rotundicauda* is present in waters with lower pH levels compared to seawaters of normal pH level (around 8.2). This corresponds well with previous findings suggesting that pH levels found in horseshoe crab habitat are low due to the presence of estuaries (mixing of freshwater with pH levels of 7–7.5), photosynthesis by algae, as well as anoxic decomposition of organic material (Jawahir et al., 2017).

Tachypleus gigas

The distribution of *T. gigas* found in this study is consistent with that found by Sekiguchi (1988), with the exception that we also found occurrences of *T. gigas* along the east coast of the Thai-Malay Peninsula (**Figure 4**). Our high-resolution projections show limited suitable habitat at the southern extension range of the species (i.e., around Java), but the low-resolution projections clearly indicate suitable habitat along the northern coastline of Java.

As with *C. rotundicauda*, we suggest that *T. gigas* is present along the coast of Myanmar even though occurrence data are missing, due to the lack of research in the area. The presumed presence of *T. gigas* in this area needs to be confirmed with real observations in the future. Alternatively, there is the possibility that other factors not included in the models, such as biological interactions or substrate conditions, may actually refrain *T. gigas* form inhabiting this region. Our models predict additional future suitable habitats in more eastern regions of Indonesia, but again we argue that *T. gigas* is unable to disperse into these areas because of the deep waters east of the continental Sunda Shelf.

T. gigas is known to occur in areas with lower salinities than our results show (15–26 PSU) (Cartwright-Taylor et al., 2011). However, *T. gigas* is not as tolerant to low salinities as *C. rotundicauda* (Chatterji, 1999), and it has been shown that salinities lower than 20 PSU prevent its eggs from hatching (Zaleha et al., 2011). Apart from salinity, the ecological tolerances of *T. gigas* found in this study are similar to those of *C. rotundicauda*, which reflect their overlapping distributions.

Although we found overlapping niches and distributions for *C. rotundicauda* and *T. gigas*, the local habitat preferences are different between the species on a smaller spatial scale, i.e., *C. rotundicauda* is associated with mangroves and mudflats and *T. gigas* with sandy sediments at more exposed beaches, and adding an environmental factor such as sediment characteristics to our models might separate the niches of the two species (Jawahir et al., 2017).

Tachypleus tridentatus

In contrast to the *T. tridentatus* distribution reported by Sekiguchi (1988), we found no recent occurrence data on this species at the coasts of Sumatra, Sulawesi, South Vietnam, or Luzon, Philippines (**Figure 5**). However, we found several occurrences along the coast of South China. Although the coast of Myanmar is an area of suitable habitat for *T. tridentatus*, we would not expect it to occur here, as the area lies far west from any sampled occurrence sites. But according to our high-resolution model, the species might occur at the coast of Sumatra, as suggested by Sekiguchi (1988), even though we did not find any recent records of its presence here. There are no occurrences of *T. tridentatus* within the Yellow Sea, and we were unable to find historical evidence suggesting previous presence here, although the species is distributed at the coasts of the Korean Peninsula and Japan as well as in the East and South China Sea. Our high-resolution models indicated suitable habitat in the Yellow Sea, but this was not confirmed by the low-resolution models, leaving some degree of uncertainty in our projections for this area. The reason why *T. tridentatus* may be absent in the Yellow Sea could be related to a variety of factors, e.g., unsuitable sediment type or breeding habitat. If this discontinuity reflects a true distribution gap, it would suggest the presence of allopatric populations of *T. tridentatus* along each side of the Yellow Sea and future studies should aim to find more evidence for a distribution gap in this area. The future prediction of *T. tridentatus* shows less suitable habitat in several areas, however, none of these areas lie within the current known distribution of the species.

All Asian horseshoe crab species are known to occur at lower salinities than reported by this study, including *T. tridentatus*, as juveniles have been found at salinities of 8–24 PSU at a nursery beach in Hong Kong (Chiu and Morton, 2004). The temperature range of *T. tridentatus* is wider than of the two other Asian species, which reflects the wider latitudinal range of *T. tridentatus,* however, ecdysis in juveniles is halted at temperatures below 22°C (Lee and Morton, 2005). Thus, temperature is likely an important factor limiting the distribution of *T. tridentatus* to the north. Our results show that *T. tridentatus* experiences a high level and narrow range of pH, which might be explained by the interaction between temperature and pH. Lower temperatures result in higher pH (Zumdahl and Zumdahl, 2000), and thus the colder waters around Japan might have a higher pH compared to warmer waters surrounding more southern Asian countries. The chlorophyll A levels found for *T. tridentatus* does not differ from the two other Asian horseshoe crabs. Our results indicate that chlorophyll A concentration, in addition to

temperature, is an important ecological factor determining the distributions of the Asian horseshoe crabs. This is supported by Hsieh and Chen (2009), who found that high densities of juvenile *T. tridentatus* were correlated with high chlorophyll A levels and high densities of polychaetes. They argued that chlorophyll A concentration reflected the amount of microalgae and hence the abundance of food available for higher trophic levels including polychaetes and that it was likely that polychaetes were an important food source for the juvenile horseshoe crabs. Thus, food resources for juvenile *T. tridentatus* increase with chlorophyll A concentration.

Potential Areas for Protection of Horseshoe Crabs

We suggest four criteria for identifying areas apt for conservation of horseshoe crabs in general, inspired from previous conservation studies (Petit et al., 1998; Hannah et al., 2007; Botero-Delgadillo et al., 2012). Marine Protected Areas (MPAs) with focus on horseshoe crab conservation should be prioritized in (1) geographic regions with disconnected distribution ranges, indicating physically distinct (i.e., allopatric) populations; (2) regions separated by large population breaks known from previous population genetic or phylogeographic studies; (3) areas with overlapping distributions of two or more species of horseshoe crabs; and (4) areas where the species experience little loss of suitable habitat under future climate scenarios. As previously noted, the potential distribution of suitable habitat, as well as the future potential distribution, found in this study should be considered as hypotheses of where Asian horseshoe crabs can be found and not as definitive results (Jarnevich et al., 2015). Thus, further investigations and models of the relationship between the species and the areas suggested here should be conducted prior to making decisions regarding conservation.

Our results show a wide gap in suitable habitat between *C. rotundicauda* populations in the southern and northern parts of the Vietnamese coast (**Figure 3A**). This, as well as the lack of mangrove habitat in the region (Hong and San, 1993), suggests the existence of allopatric populations of *C. rotundicauda* on the southern and northern coasts of Vietnam. Additionally, there is a gap in the distribution of suitable habitat at the northeast coast of Peninsular Malaysia, and there is evidence that genetically distinct populations of *C. rotundicauda* reside on each side of the Thai-Malay Peninsula (Obst et al., 2012; Adibah et al., 2015). The suitable habitat distribution also shows two separate distribution areas of *C. rotundicauda* on Borneo and Java, which could be allopatric to the three distinct distribution sites at mainland Southeast Asia. Consequently, based on our models, we hypothesize that there are probably five separated distribution ranges of *C. rotundicauda* in Southeast Asia, i.e., India to South Peninsular Malaysia, East Thailand to South Vietnam, North Vietnam to South China, Java, and Borneo, and none of these areas will become less suitable in the future according to our study. Based on our criteria 1, 2, and 4, we suggest the establishment of MPAs in coastal areas of West and East Thailand, North Vietnam and South China, and Borneo, in order to conserve *C. rotundicauda* in Southeast Asia (**Figure 6**).

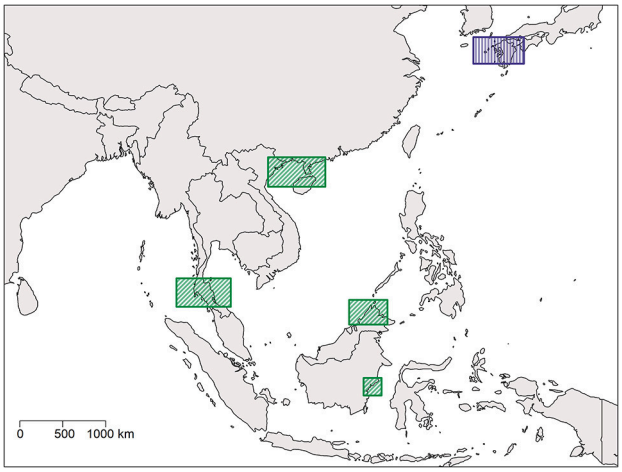


FIGURE 6 | Map of Southeast Asia and Japan showing regions which are optimal for the establishment of MPAs in order to conserve the three Asian species of horseshoe crabs. Rectangles with diagonal lines and green color indicate areas where MPAs for horseshoe crab conservation are absent, whereas rectangles with vertical lines and purple color indicate areas where MPAs for horseshoe crab conservation are already established.

Contrary to *C. rotundicauda*, we do not expect allopatric populations of *T. gigas* to exist across the Thai-Malay Peninsula as there is a continuity of suitable habitat for *T. gigas* in this region (**Figure 4A**). This assumption is supported by the short genetic distance between *T. gigas* collected from Vietnam and the Andaman Sea (Obst et al., 2012). The contrast in habitat continuity between *T. gigas* and *C. rotundicauda* can be explained by different habitat preferences for the two species. *C. rotundicauda* is more confined to mangroves and river deltas, while *T. gigas* is more frequent in truly marine environments, which is likely to result in higher dispersal along the coast. However, there is evidence of limited dispersal between the *T. gigas* populations in Northwest and Southwest Peninsula Malaysia (Rozihan and Ismail, 2011). Thus, based on our results, we hypothesize three distinct distribution areas for *T. gigas*: India to South Vietnam, Java, and Borneo. Additionally, *C. rotundicauda* and *T. gigas* are co-occurring at several locations along the coasts of the Southeast Asian mainland, hence according to criteria 1, 3, and 4 we suggest establishment of MPAs at the locations along the Southeast Asian mainland as wells as Borneo (**Figure 6**).

For *T. tridentatus* our results show four distinct distribution ranges; Japan to South Korea, China to North Vietnam, Borneo to Philippines, and Java (**Figure 5A**). *T. tridentatus* has overlapping distribution ranges with the two other Asian species; *T. tridentatus* and *C. rotundicauda* co-occur in South Vietnam as well as in the Hainan and Guangxi region of the Chinese coast, and all three Asian species co-occur along the coasts of Borneo and Java. According to criteria 1, 3, and 4, establishment of MPAs should hence be recommended in Japan, North Vietnam and South China, and Borneo (**Figure 6**). In Japan, MPAs are already established (http://mpatlas.org/explore/), and some of these include important breeding sanctuaries for horseshoe

crabs, such as the Saikai National Park in Nagasaki. For areas where *C. rotundicauda* occurs sympatrically with *T. gigas* or *T. tridentatus*, it is important to consider the different habitat preferences for the species, i.e., low-salinity mangroves and river estuaries for *C. rotundicauda*, and high salinity sandy beaches for *T. gigas* and *T. tridentatus*.

Based on our analysis we propose that at least five regions should have high priority for horseshoe crab conservation: four in Southeast Asia, based on geographically separated areas with suitable habitat as well as the co-occurrence of two or more species, and one in Japan for the protection of distinct populations of *T. tridentatus* (**Figure 6**). We recommend that future MPAs are examined on site to ensure their suitability and that specific habitats, i.e., mangroves, mudflats, or gentle slope beaches, are present in order to provide the appropriate habitat for the horseshoe crab species in question (e.g., Kwan et al., 2016), as some factors of potential importance for horseshoe crab habitat suitability, such as sediment characteristics and presence of anthropogenic structures, could not be included in our analysis. The work presented here provides an initial step in the conservation of the Asian horseshoe crabs species, a research area in need of increasing attention considering the importance of these coastal organisms to human subsistence and health.

Future Studies

Destruction of breeding habitats pose a significant threat to horseshoe crabs (e.g., Mishra, 2009; Zaldívar-Rae et al., 2009; Nelson et al., 2016) and adult horseshoe crabs are harvested on the beaches during spawning for commercial exploitation. Since our study focused on the distribution range of adult and juvenile horseshoe crabs, we can only draw few conclusions about the distribution and preferred environmental conditions of horseshoe crab breeding sites. However, the approach implemented in this study could also be used to inventory and predict potential breeding sites using scientific networks and fieldwork, but should then include additional environmental variables. Horseshoe crabs nest on beaches with slight slopes and in mangroves and estuaries. Hence data on habitat types, beach inclination, sediment type and granularity, wave action, and organic contents of the sediment are essential variables that need to be measured and included for such analysis. Additionally, our study includes tidal regime environmental variables in the ecological niche modeling that can be used in future research on the distributions of other intertidal organisms, and thus could be of importance for the conservation of intertidal communities at a global scale.

CONCLUSIONS

The distributions of the three Asian horseshoe crab species have been assembled with this work, and marine areas of suitable habitat have been identified. Additionally, our models predict that *C. rotundicauda* and *T. gigas* in the future will experience increased habitat suitability. The information on ecological variables important for the three species, as well as the current and future distributions of suitable habitat provided here, can be used as an initial step in determining where MPAs should be established for conservation purposes and to determine the IUCN conservation status of the three Asian horseshoe crab species.

AUTHOR CONTRIBUTIONS

SV, PF, and MO: designed the study; SV, MO, II, and PF: collected the data; MO and FQ: performed the modeling and analysis; SV: led the writing of the first draft in close collaboration with PF and MO. All authors contributed substantially to the revisions.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2018.00164/full#supplementary-material

Supplementary Data Sheet 1 | Explanation of the construction of global tidal variables.

Supplementary Image 1 | Heat map for *C. rotundicauda* using Maxent.

Supplementary Image 2 | Heat map for *C. rotundicauda* using SVM.

Supplementary Image 3 | Heat map for *T. gigas* using Maxent.

Supplementary Image 4 | Heat map for *T. gigas* using SVM.

Supplementary Image 5 | Heat map for *T. tridentatus* using Maxent.

Supplementary Image 6 | Heat map for *T. tridentatus* using SVM.

Supplementary Table 1 | Data provider names for occurrence records obtained by own observations, collaborators, and scientific network.

Supplementary Table 2 | Data references for occurrence records obtained by literature.

Supplementary Table 3 | Data provider names for obtained GBIF occurrence records. The name (or acronym) in use by the institution having custody of the object(s) or information referred to in the record.

Supplementary Table 4 | Test statistics for analysis of differences in environmental conditions between four horseshoe crab species using Welch's test for unequal variances.

Supplementary Table 5 | Multiple comparisons of difference between means of environmental variables for four horseshoe crab species (CR – *C. rotundicauda*,

TG – *T. gigas*, TT – *T. tridentatus*) using Games-Howell *Post-Hoc* test. Significant mean difference at the 0.05 level in bold.

Supplementary Table 6 | Loadings for the different environmental variables shown for all PCA components for *C. rotundicauda*.

Supplementary Table 7 | Loadings for the different environmental variables shown for all PCA components for *T. gigas*.

Supplementary Table 8 | Loadings for the different environmental variables shown for all PCA components for *T. tridentatus*.

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Area-Independent Effects of Water-Retaining Features on Intertidal Biodiversity on Eco-Engineered Seawalls in the Tropics

Lynette H. L. Loke[1], Eliza C. Heery[1], Samantha Lai[1], Tjeerd J. Bouma[2,3] and Peter A. Todd[1]*

[1] Experimental Marine Ecology Laboratory, Department of Biological Sciences, National University of Singapore, Singapore, Singapore, [2] Department of Estuarine and Delta Systems, Royal Netherlands Institute for Sea Research (NIOZ), Utrecht University, Yerseke, Netherlands, [3] Faculty of Geosciences, Department of Physical Geography, Utrecht University, Utrecht, Netherlands

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***Correspondence:**
Lynette H. L. Loke
lynetteloke@gmail.com;
dbsloke@nus.edu.sg

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Over the last decade there has been a global effort to eco-engineer urban artificial shorelines with the aim of increasing their biodiversity and extending their conservation value. One of the most common and viable eco-engineering approaches on seawalls is to use enhancement features that increase habitat structural complexity, including concrete tiles molded with complex designs and precast "flowerpots" that create artificial rock pools. Increases in species diversity in pits and pools due to microhabitat conditions (water retention, shade, protection from waves, and/or biotic refugia) are often reported, but these results can be confounded by differences in the surface area sampled. In this study, we fabricated three tile types (n = 10): covered tile (grooved tile with a cover to retain water), uncovered tile (same grooved tile but without a cover) and granite control. We tested the effects of these tile types on species richness (S), total individual abundance (N), and community composition. All tiles were installed at 0.5 m above chart datum along seawalls surrounding two island sites (Pulau Hantu and Kusu Island) south of Singapore mainland. The colonizing assemblages were sampled after 8 months. Consistent with previous studies, mean S was significantly greater on covered tiles compared to the uncovered and granite tiles. While it is implied in much of the eco-engineering literature that this pattern results from greater niche availability allotted by microhabitat conditions, we further investigated whether there was an underlying species-individual relationship to determine whether increases in S could have simply resulted from covered tiles supporting greater N (i.e., increasing the probability of detecting more species despite a constant area). The species-individual relationship was positive, suggesting that multiple mechanisms are at play, and that biodiversity enhancements may in some instances operate simply by increasing the

abundance of individuals, even when microhabitat availability is unchanged. This finding underscores the importance of testing mechanisms in eco-engineering studies and highlights ongoing mechanistic uncertainties that should be addressed to inform the design of more biodiverse seawalls and urban marine environments.

Keywords: habitat complexity, ecological engineering, urban marine ecology, species-area relationship, species-abundance relationship

INTRODUCTION

Growing coastal populations that are increasingly threatened by rising seas (Hinkel et al., 2014; Nicholls, 2015) are driving the construction of coastal defenses and flood protection, particularly in low-elevation coastal zones and vulnerable urban centers (Neumann et al., 2015). Ideally, coastal adaptation planning and defense systems should employ nature-based approaches, including "planned retreat" (pulling developments back from the present shoreline) and the creation or restoration of ecosystems that naturally protect against flooding, such as salt marshes and mangroves (Temmerman et al., 2013; Narayan et al., 2016). However, traditionally engineered artificial structures, such as seawalls, are pervasive, particularly in coastal cities, where sea level rise adaptation efforts primarily aim to protect the existing shoreline or advance the shoreline into adjacent marine habitats (Nicholls, 2011; Dafforn et al., 2015). These structures have considerable impacts on marine ecosystems (Bishop et al., 2017; Heery et al., 2017, 2018), yet are likely to remain a central component of coastal defense systems, either as the primary mode of flood protection or in combination with natural features (Bulleri and Chapman, 2009; Cheong et al., 2013). In response, there has been a global effort during the last decade to ecologically engineer hard artificial structures with the aim of improving their ecological value and alignment with conservation goals (Chapman and Underwood, 2011; Dyson and Yocom, 2015; Pioch et al., 2018). Ecological engineering solutions have already been applied to artificial shoreline projects in several cities, with various benefits to both marine ecosystems and human communities in urban areas (Arkema et al., 2017; Morris et al., 2018). Yet, for many of them, there have been few experimental studies that systematically explore precisely *how* they alter marine biota. A thorough understanding of the underpinning mechanisms through which ecological enhancements operate remains elusive.

Though ecological engineering of artificial structures can be used to meet a variety of objectives, its core environmental aims are often to enhance biodiversity and extend the conservation of native marine species to urban habitats that would otherwise be inhospitable (Loke et al., 2019). One of the most popular techniques for intertidal seawalls is to increase habitat structural complexity (Loke et al., 2014, 2017), for instance by removing pieces of the seawall to create recesses (Chapman and Blockley, 2009), drilling pits to form rock pools (Martins et al., 2010; Evans et al., 2016; Hall et al., 2018), and attaching "flower pots" and other concave enhancements (Browne and Chapman, 2011, 2014; Firth et al., 2016a; Waltham and Sheaves, 2018). These manipulations have repeatedly been shown to increase the richness and abundance of benthic organisms on intertidal seawalls (reviewed by Loke et al., 2019), and their effectiveness is frequently attributed to increased shade and moisture (i.e., increased niche availability) allotted by structurally complex microhabitat features (Firth et al., 2014, 2016b; Perkol-Finkel et al., 2018; Strain et al., 2018), which can reduce temperature fluctuations, minimize desiccation stress, and facilitate the recruitment of sessile fauna (Metaxas and Scheibling, 1993; Blockley and Chapman, 2006; Seabra et al., 2011; Firth et al., 2013). To date, the majority of related experiments in this area have been conducted in temperate areas (e.g., Browne and Chapman, 2011; Evans et al., 2016; Hall et al., 2018). The deleterious effects of temperature and desiccation are potentially a greater threat to tropical intertidal assemblages, and hence more data are needed from low latitude countries.

Discerning between the mechanisms that explain why structurally complex features enhance intertidal diversity is essential for ensuring that ecological engineering achieves its objectives and that unintentional consequences are avoided, particularly as these features are now being "scaled-up" to meet growing commercial demand for ecological enhancements. As has been widely suggested in past literature, creating additional microhabitat features may indeed increase biodiversity primarily by increasing niche availability. However, this has rarely been tested explicitly and various other explanations are often feasible. For instance, greater surface area in structurally complex treatments will lead to greater species richness (due to basic species-area relationships, SAR) and prove misleading if not carefully accounted for in eco-engineering studies. Alternatively, even when surface area is accounted for, an increase in niche availability is just one of the possible mechanisms through which enhanced biodiversity can arise. For example, microhabitat conditions associated with structurally complex features may instead support greater abundances/densities of individual organisms, which in turn could increase species richness simply by increasing the probability that more species will be detected as more individuals are sampled (henceforth referred to as "species-individual relationship"; see Fisher et al., 1943).

In this study, we explored multiple alternative explanations for biodiversity increases that have been well-documented in response to structurally complex eco-engineering features and informally linked to increases in niche availability. To determine whether, in fact, changes in microhabitat conditions (e.g., combined effect of increased water retention, shade, protection from waves and/or refugia) provide a parsimonious explanation of increased species richness on structurally complex features, we accounted for the potentially confounding effect of surface area on species richness in our statistical analyses. Furthermore,



FIGURE 1 | Photographs (taken 6 months after deployment) of the three tile types tested in this study: **(A)** covered tile, **(B)** uncovered tile, and **(C)** granite control tile. Note the underlying tile design and surface area of **(A)** and **(B)** are the same.

to discern whether microhabitat conditions primarily altered biodiversity by increasing the availability of niches, or whether other non-niche-related mechanisms are also important, we conducted an *a posteriori* analyses of the relationship between the number of individuals (*N*) and the number of species (*S*).

We deployed tiles of the same underlying tile design (and thus level of topographic complexity) to seawalls in Singapore—treatments differed only in whether they had water retaining features (i.e., had a cover). Consistent with previous findings, we hypothesized that: (1) Covered tiles would host greater species richness (*S*) compared to uncovered and granite (control) tiles; (2) Covered tiles would host a greater number of individuals (total abundance, *N*) compared to uncovered and granite tiles; and (3) Covered tiles would host assemblages that were compositionally different compared to uncovered and granite tiles. We additionally examined the species-individual relationship across treatments to assess whether there was evidence that microhabitat (niche) availability was likely the primary mechanism underlying these patterns. Note that by exploring the relationship between *S* and *N*, we did not explicitly test any mechanism *per se*, rather, we used this analysis to rule out the possibility that increases in *S* could be explained by increases in *N*, a pattern that would suggest that niche availability is not the only mechanism influencing biodiversity when surface area is held constant.

MATERIALS AND METHODS

Experimental Design, Tile Fabrication and Deployment

We fabricated three tile types: "covered" tile (grooved tile with a concrete cover plate to retain water, **Figure 1A**), "uncovered" tile (same grooved tile but without a cover, **Figure 1B**) and "granite" tile (**Figure 1C**). The concrete cover plate served to trap water and provide shade and protection from waves and predators in the "covered" tile treatment, but these components were not disentangled by our study. All tiles measured 200 × 200 × 60 mm (width × length × depth). The covered and uncovered tiles were made of concrete following the "complex-groove quarter tile" design used in Loke and Todd (2016) and were cast from silicone rubber molds using a 1:3 Portland cement to sand mix. Granite tiles, which served as a control, were made of broken granite slabs cemented onto a concrete base to mimic the surface of the surrounding granite seawall (for more details on how the tiles were constructed please refer to Loke and Todd, 2016). This was a procedural control and provided an idea of what the background seawall would support within the same timeframe given the same mounting system as the concrete tiles.

The tiles were deployed during low tides on 9–10 August 2010 along un-grouted, sloping (≈30°), granite rip-rap seawalls at two sites south of Singapore's mainland: Pulau Hantu (1° 13′ 34″ N, 103° 45′ 0″ E) and Kusu Island (1° 13′ 22″ N, 103° 51′ 40″ E). Ten replicates of each tile type were attached onto the seawalls in random order using M8 stainless steel bolts (**Figure 1**). All tiles were installed flat along the sloping seawall (see **Figure 1**) 0.5 m above chart datum and spaced at least 2.0 m apart. Hence, there was a total of 60 tiles with "tile type" as a fixed factor and "site" as a random factor; i.e., 2 sites × 3 tile types × 10 replicates.

Field Sampling and Laboratory Procedures

The tiles were retrieved after 8 months. The colonizing benthic assemblage on the tiles was sampled; organisms on tile surfaces were scraped into trays immediately after retrieval. Benthic organisms colonizing the outside of the concrete cover plate on the covered tiles were not included, but those on the underside were. Even though the concrete plates provided additional substrate where they covered the grooves of the underlying tile, they also removed habitable area where they were in contact with the top of the ridges. The overall effect was that the covered tiles had slightly less surface area (60 cm^2) available for colonization than the uncovered tiles. This amounts to a 6.9% difference that was accounted for in our statistical models (see section "Statistical Analyses"). Note that, although water retained by the covered tiles presumably provided greater three-dimensional habitat for suspended organisms, these organisms were lost when covers were removed and thus were not among the organisms sampled. All specimens were stored in 70% ethanol until they were manually sorted, counted and identified to species or morphospecies level (within known genera) except for polychaetes and amphipods, which were identified to family level, and algae, which were identified to functional group level following Loke et al. (2016).

TABLE 1 | Estimated parameters, standard errors, *z*-values, and *p*-values for the negative binomial mixed effects models; significant *p*-values are in bold.

Models	Estimate	SE	*z*-value	*p*-value
Species richness (*S*)				
Intercept	3.52	0.09	38.49	**<0.001**
Uncovered tile	−2.09	0.10	−20.35	**<0.001**
Covered tile	−0.50	0.10	−5.25	**<0.001**
Number of individuals (*N*)				
Intercept	5.68	0.85	6.72	**<0.001**
Uncovered tile	−2.02	0.17	−11.87	**<0.001**
Covered tile	−0.49	0.17	−2.85	**<0.01**

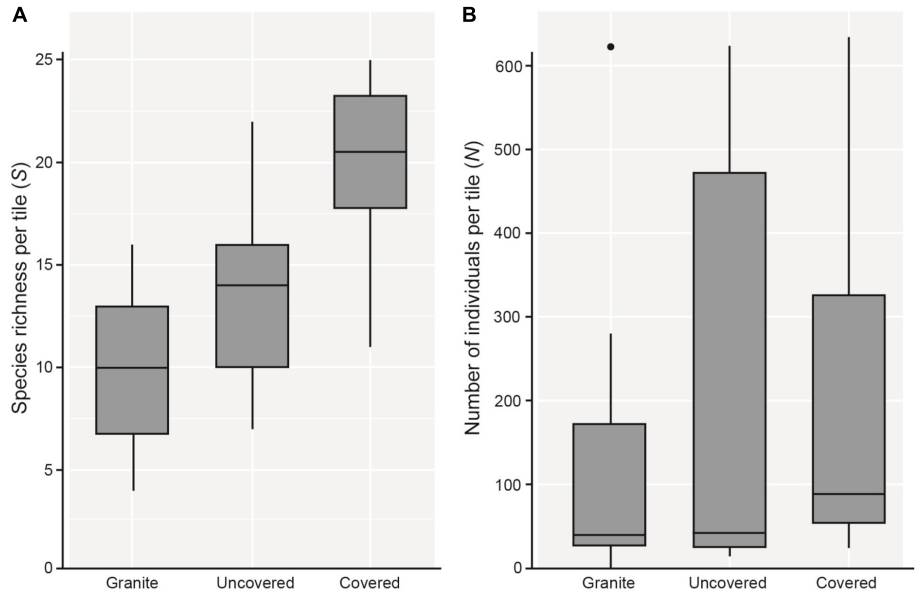


FIGURE 2 | Boxplots showing 5 and 95% confidence intervals (whiskers), 25 and 75% confidence intervals (boxes), and median of **(A)** species richness (*S*), and **(B)** total number of individuals (total abundance; *N*) found on each tile type.

Statistical Analyses

As some tiles were lost due to wave action, there were six covered tiles, seven uncovered tiles and six granite tiles remaining at Kusu Island, resulting in an unbalanced ANOVA design (no tiles were lost at Pulau Hantu). To avoid removing a large number of replicates to run a balanced analysis, we modeled species richness (*S*) and number of individuals (*N*) as counts using mixed effects models with negative binomial error terms to test for differences in *S* and *N* among tile types (Hypotheses 1 and 2, respectively). We selected the negative binomial error structure due to overdispersion in the Poisson models and included surface area as an offset in the model (Zuur et al., 2009). All models were constructed in R version 3.3.0 (R Development Core Team, 2016) with tile type as a fixed effect and site as a random effect. We then explored the potential species-individual relationship (i.e., relation between *S* and *N*) via simple linear regression [*lm()* function in R]. Following the protocols outlined in Zuur et al. (2010), prior to our analyses we examined the data for overdispersion, non-linearity, heteroscedasticity, and presence of outliers.

To test for differences in community composition between tile types (Hypothesis 3), we used permutational distance-based multivariate analysis of variance (PERMANOVA; Anderson, 2001) to analyze the full resemblance matrix calculated on Bray-Curtis similarities from log-transformed abundances with site as a random factor and tile type as a fixed factor. Due to highly significant differences between the communities among the sites, we then ran separate PERMANOVAs for each site. *P*-values were generated using 9999 unrestricted random permutations of residuals. Canonical analysis of principal coordinates (CAP) was applied to examine whether the communities on the three tile types could be separated and to explore the potential relationships between the individual species found and the canonical axes. The significance of the canonical relationship in each combination was tested using 9999 unrestricted random permutations of the transformed community data (Anderson et al., 2008). All multivariate analyses were performed using the PERMANOVA+ add-on for PRIMER v6 (Anderson et al., 2008).

RESULTS

A total of 8925 individuals representing 52 different faunal species and six algal functional groups were recorded. None of these species are unique to seawalls in Singapore (i.e., they are also found on natural rocky shores in Singapore; Lai et al., 2018) and none are known to be non-native (Jaafar et al., 2012; Tan et al., 2018).

As hypothesized, covered tiles hosted significantly greater *S* (χ^2 = 504.61, df = 2, $p < 0.001$) and *N* (χ^2 = 153.65, df = 2, $p < 0.001$) compared to uncovered and granite tiles (**Table 1** and **Figure 2**). We also found that the variability in *N* was due primarily to the dominance by a single species: *Siphonaria guamensis* at Kusu island (**Figure 3**). *S. guamensis* (the false limpet) was generally absent on tiles at Pulau Hantu (only a single individual recorded) but present in disproportionately large numbers at Kusu Island (319.1 ± 154.8 individuals per tile; mean ± SD) (**Figure 3**).

A plot of the abundance of *S. guamensis* alone revealed a similar trend to the plot of total individual abundance summarizing the whole dataset (i.e., **Figure 3A** vs. **Figure 2B**). On removing *S. guamensis*, the *N* pattern resembled the *S* pattern, with the greatest number of individuals found on covered

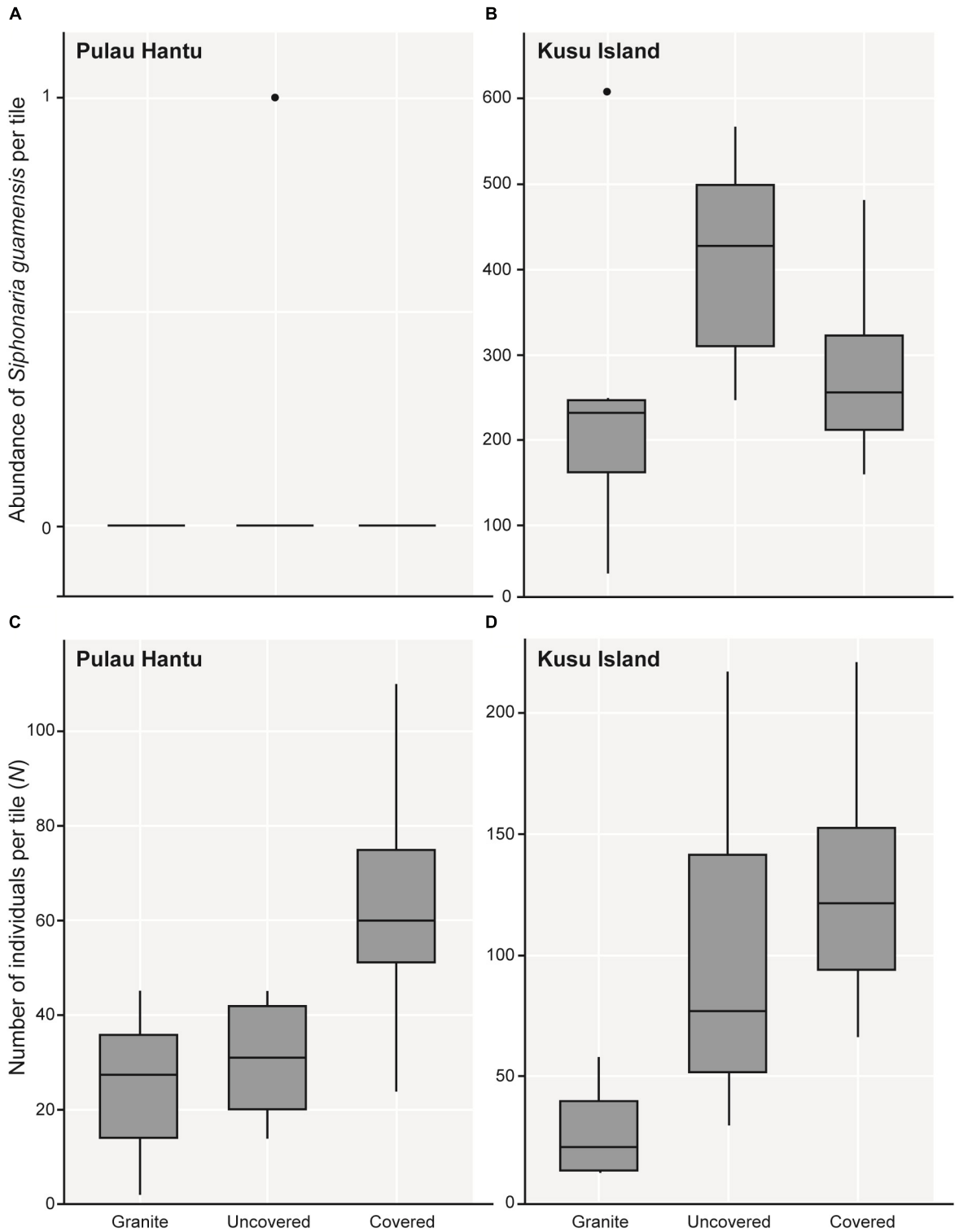


FIGURE 3 | Boxplots showing 5 and 95% confidence intervals (whiskers), 25 and 75% confidence intervals (boxes), and median of the total abundance of **(A)** *Siphonaria guamensis* in each tile treatment at Pulau Hantu and **(B)** Kusu Island; **(C)** the total number of individuals (*N*) of all 51 species in each tile treatment at Pulau Hantu and **(D)** Kusu Island after removing *S. guamensis*.

tiles, followed by uncovered and granite tiles (**Figures 3B–C** vs. **Figure 2A**).

The simple linear regression between *S* and *N* revealed an overall significant positive relationship although the trend at each site differed, and was apparently non-linear at Kusu Island, again due to the disproportionately large abundances of *S. guamensis* (**Figures 4A–B**). As with the mixed effects model for total abundance, the removal of *S. guamensis* revealed highly significant positive linear relationships between *S* and *N* at both sites (**Figures 4C–D**).

Due to significant community differences between the assemblages at Pulau Hantu and Kusu Island (PERMANOVA; df = 1, 43, MS = 34172, Pseudo-F = 26.3, P (perm) < 0.001) we performed separate PERMANOVAs for each site. Results revealed that community composition differed significantly among the tile types at each site except between "uncovered" and "granite" tiles at Pulau Hantu (**Table 2**). The CAP analysis also showed a significant separation between the tile types at each site ($p < 0.05$; **Figure 5**). Species commonly found under rocks, such as *Barbatia amygdalumtostum* and *Isognomon legumen*, characterized covered tiles.

DISCUSSION

Many of the ecological engineering efforts to improve biodiversity on urban structures have employed the use of water-retaining features (Firth et al., 2016a), but these features often differ in surface area from less structurally complex controls, and could be influencing species numbers. Without controlling for area, it is difficult to attribute increases in *S* solely to changes in microhabitat conditions—e.g., water retention, shade, protection from waves and/or biotic refugia—as species-area effects can dominate, as has been shown across nearly all systems and scales (Fahrig, 2013). Our study demonstrates experimentally that increased *S* from water-retaining features is indeed a function of microhabitat provisioning (i.e., niche availability) to some extent, and this effect is independent of surface area. In fact, even though covered tiles had 6.9% less surface area than uncovered tiles, they supported greatest *S*. By ruling out surface area as the driver of enhanced biodiversity, this finding adds an essential piece to the mechanistic puzzle underpinning ecological enhancements of shorelines, and strengthens the foundation of ecological engineering as a discipline, ultimately aiding in its advancement.

We ruled out surface area as the driver of diversity patterns by comparing covered and uncovered tile treatments that offered the same structurally complex substrate for benthic organisms. As in previous studies, structurally complex tiles (both covered and uncovered tile treatments) were found to increase abundance and species richness compared to the less complex substrata that characterize seawalls (Loke and Todd, 2016). However, the added value from this study is exemplified by the differences observed between covered versus uncovered treatments. We observed higher *S* on covered tiles compared with uncovered tiles. Covered tiles also supported greater *N*, and this effect was especially pronounced when the disproportionately abundant false limpet, *S. guamensis*, was removed (discussed further below). These findings are consistent with past studies (e.g., Chapman and Blockley, 2009; Browne and Chapman, 2011; Waltham and Sheaves, 2018), and suggest that incorporating additional microhabitats (e.g., shade and water retaining features) may be an effective strategy for biodiversity enhancement. More broadly, the findings lend weight to the idea that enhancing habitat (niche) complexity can and does play an important role in supporting more diverse communities on seawalls.

Water-retaining features are thought to enhance the diversity of intertidal biota through multiple mechanisms. Pits/pools are continually submerged, eliminating desiccation stress and other risks associated with emersion during low tide (Connell, 1972), and are rapidly colonized by a suite of taxa (Underwood and Jernakoff, 1984). They also provide greater shade than emergent surfaces, which dampens temperature variability (Seabra et al., 2011), reduces thermal stress (Williams, 1994; Bertness et al., 1999), and provides highly localized refugia for temperature-sensitive taxa, such as gastropods (Garrity, 1984) and newly recruited sessile invertebrates (Blockley and Chapman, 2006; Chapman and Blockley, 2009). This can lead to net increases in diversity (Blockley, 2007), even though reduced irradiance is limiting for some taxa, particularly primary producers (Goldberg and Foster, 2002). Mediated desiccation and temperature stress act in combination with a complex suite of additional abiotic and biotic variables (e.g., dissolved gas accumulation and decreasing pH during low tide, Huggett and Griffiths, 1986) to influence the composition of organisms in pits/pools, and community response to water-retaining features is therefore highly variable (Bugnot et al., 2018). The addition of a cover plate may also

TABLE 2 | PERMANOVA results based on Bray–Curtis dissimilarities of the relative abundances (pre-treated with log[X + 1] transformation) of 56 different species in response to the tile type treatments.

Source	df	MS	Pseudo-*F*	*p* (perm)	Unique perms
Pulau Hantu					
Tile type	2	3666	2.1274	**0.0084**	9913
Residual	27	1723.2			
Total	29				
Kusu Island					
Tile type	2	2176.6	3.7061	**0.0001**	9923
Residual	16	587.3			
Total	18				

Groups	*t*	*p* (perm)	Unique perms
Pulau Hantu			
Covered, uncovered	1.7646	**0.0085**	9418
Covered, granite	1.779	**0.0027**	9402
Uncovered, granite	0.66005	0.9666	9384
Kusu Island			
Covered, uncovered	1.4639	**0.0023**	460
Covered, granite	2.2765	**0.001**	1708
Uncovered, granite	1.8583	**0.0085**	9418

Significant p (perm) values are highlighted in bold.

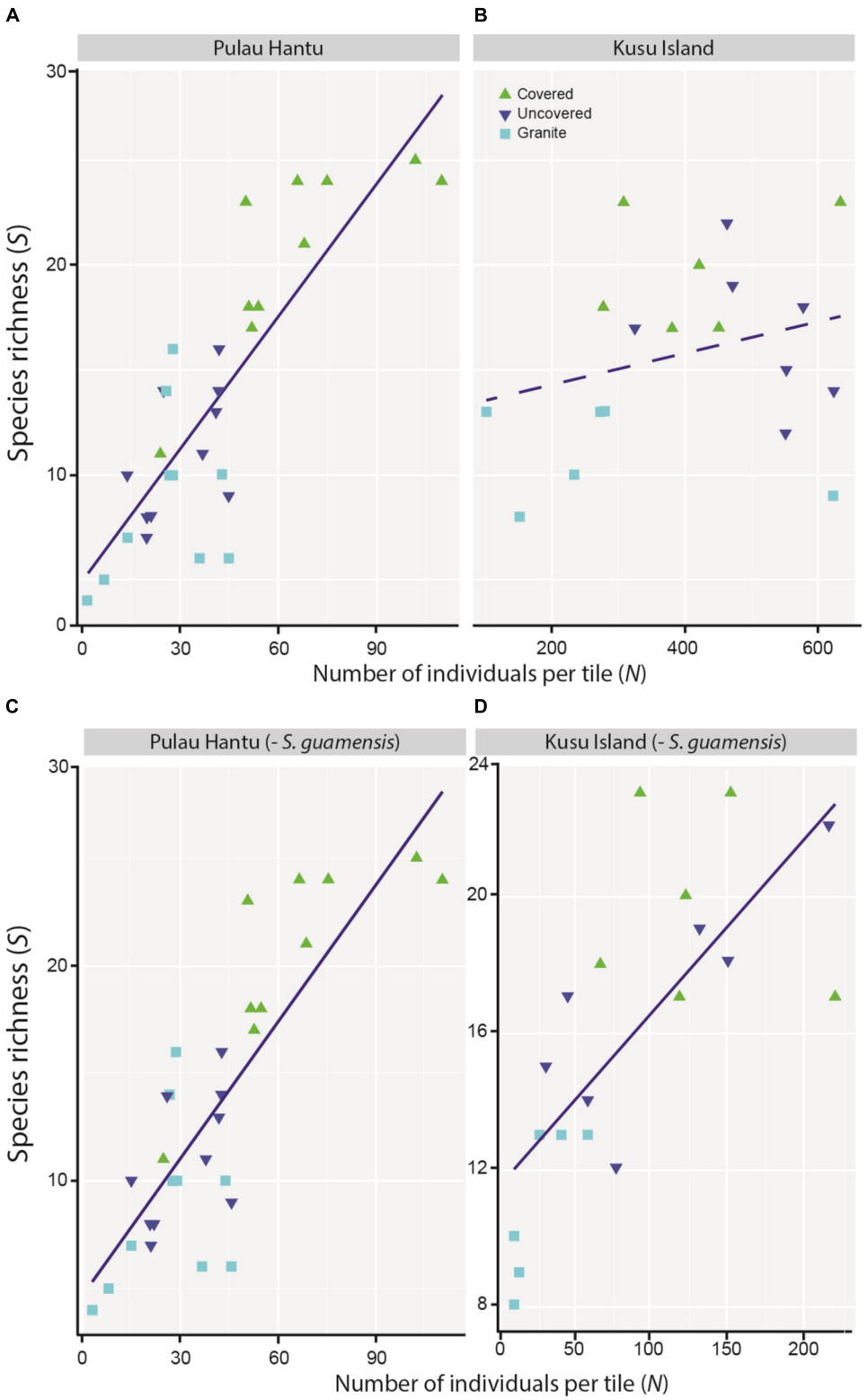


FIGURE 4 | Simple linear regression between species richness (*S*) and total number of individuals (*N*) at **(A)** Pulau Hantu (r^2 = 0.68, p = 0.000), and **(B)** Kusu Island (r^2 = 0.19, p = 0.262), and after removing *S. guamensis* at **(C)** Pulau Hantu (r^2 = 0.54, $p < 0.001$), and **(D)** Kusu Island (r^2 = 0.66, $p < 0.001$).

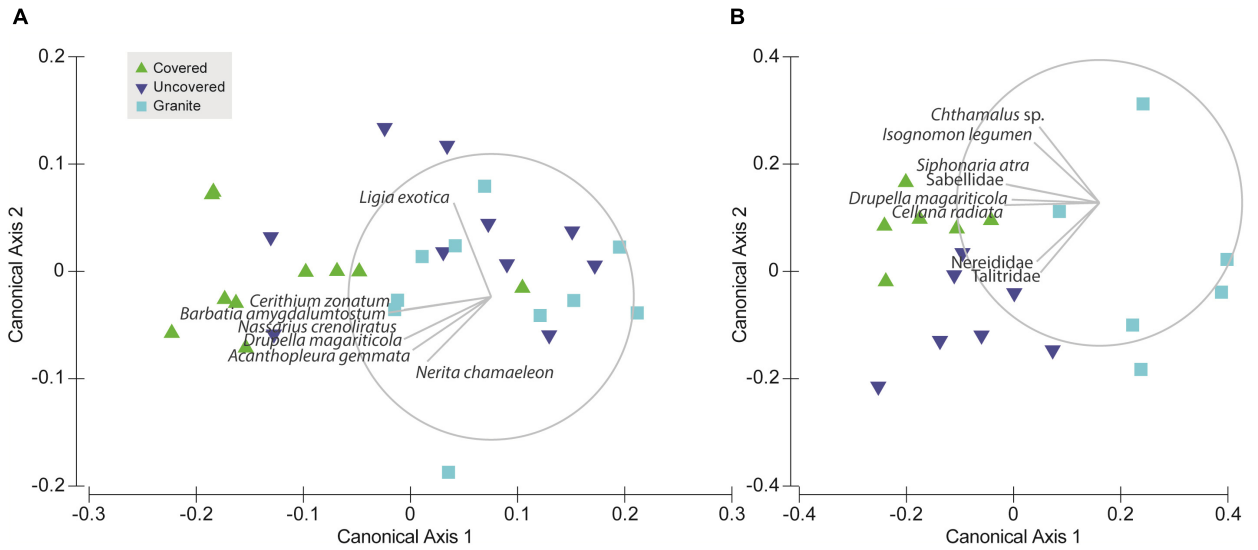


FIGURE 5 | Canonical analysis of principal coordinates (CAP) ordination for the tile treatments at each site: **(A)** Pulau Hantu and **(B)** Kusu Island; the vector overlay is applied to each plot using Pearson correlation ($\rho = 0.6$) to explore species relationships with the canonical axes.

afford organisms greater shelter from wave action and water movement—potentially a critical resource at sites with higher wave energy such as Kusu Island (Loke and Todd, 2016)—and possibly additional protection from predators.

Although discerning specific niche mechanisms was beyond the scope of this study, differential responses to tile treatments observed among intertidal fauna provides some insights into key abiotic differences between our treatments. For instance, *S. guamensis*, which was primarily found on uncovered tiles at Kusu Island, was 1–2 orders of magnitude more abundant than all other faunal species. This species tends to dominate exposed flat surfaces (Amnuaypon and Wangkulangkul, 2018; Loke pers. obs.), which may explain why uncovered tiles hosted a greater abundance of *S. guamensis* than the covered tiles (**Figure 3A**); uncovered tiles had a greater amount of "exposed area." *S. guamensis* was found to contribute significantly to the site differences in community structure but not to differences in species composition among tile types (**Figure 4**). This suggests *S. guamensis* has no strong preference for particular topographical features or substrate material compared to the other gastropods we found at Kusu Island. Conversely, *Drupella magariticola*, *Cellana radiata*, and *S. atra*, were found to be driving the community differences between the concrete tiles (both covered and uncovered tiles) and the granite tiles. This could be due to differences in both substrate material and topographic complexity. Covered and uncovered tiles at Kusu Island were distinguished along the second CAP axis by the pod tree oyster, *I. legumen*, which usually occurred in clumps on the face beneath the cover (Loke pers. obs). At Pulau Hantu, assemblages colonizing the uncovered and granite tiles were not compositionally distinguishable (**Figure 5** and **Table 2**) but it was evident that covered tiles were distinct from the granite tiles. This is mainly due to *Cerithium zonatum* and *B. amygdalumtostum* (bearded ark clams); the latter are usually found on the underside of rocks and boulders on natural shores. Thus, our study shows that, even by adding one more habitat niche (thereby increasing "informational complexity," *sensu* Loke et al., 2015), we can increase the number of species by providing habitat for specialist species with narrower niche breadths.

While our study demonstrates biodiversity increases from water-retaining features irrespective of surface area, these increases are probably not due to microhabitat differences (niche effects) alone. This is evidenced by the strong positive relationship we observed between the total number of individuals (N) and species richness (S), particularly after accounting for *S. guamensis*. Had the covered tiles captured a greater species richness but not greater individual abundance compared to the uncovered and granite tiles, it would have shown that the improved richness was due solely to the niche effects provided by the covered tile. Instead, the consistent relationship between S and N suggests that in addition to the niche effects, treatment differences might have simply resulted in the capture of more individuals, which in turn increased the likelihood of capturing more species. While this does not take away from the fact that microhabitat properties such as water retention play an important role in improving species richness, it underscores the importance of accounting for the possible influence of N on S when interpreting results from eco-engineering studies.

CONCLUSION

In conclusion, water-retaining features combined with complex topography can enhance biodiversity on existing seawalls independently of area through mechanisms that act directly via niche effects, and possibly indirectly via total individual abundance. Until we are able to discern the relative importance of these mechanisms, we suggest that they be explored

empirically (for instance, by examining species-area relationships (SARs) and/or species-individual relationships) as alternative explanations for the results of future eco-engineering studies. Such efforts are needed to ensure that claims regarding the efficacy of eco-engineering solutions to increase species diversity are justified and well supported.

AUTHOR CONTRIBUTIONS

LL, TB, and PT designed the experiments. LL conducted the experiment and collected the data. LL and EH analyzed the data. LL wrote the first draft of the manuscript. All authors contributed to the revisions.

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Filling the Data Gap – A Pressing Need for Advancing MPA Sustainable Finance

John J. Bohorquez, Anthony Dvarskas and Ellen K. Pikitch*

School of Marine and Atmospheric Sciences, Stony Brook University, Stony Brook, NY, United States

Reaching protected area (PA) coverage goals is challenged by a lack of sufficient financial resources. This funding gap is particularly pervasive for marine protected areas (MPAs). It has been suggested that marine conservationists examine examples from terrestrial protected areas (TPAs) for potential solutions to better fund MPAs. However, the funding needs for MPAs and TPAs have not been directly compared, and there is risk of management failures if any such differences are not properly considered when designing MPA financial strategies. We perform an in-depth literature review to investigate differences in distribution of costs incurred by MPAs and TPAs across three primary categories; establishment, operational, and opportunity costs. We use our findings to conduct a snapshot quantitative comparison, which we complement with theoretical support to provide preliminary insight into differences between MPA and TPA costs, and how these may influence financial strategies most appropriate for each type of PA. Our research suggests that TPA costs, and thereby funding requirements, are greater for the time period leading up to and including the implementation phase, whereas MPAs have higher financial requirements for meeting long-term annual operational costs. This may be primarily due to the prevalence of private property rights for terrestrial regions, which are less frequently in place for ocean areas, as well as logistical requirements for enforcement and monitoring in a marine environment. To cement these suggestions in greater analytical certainty, we call for more thorough and standardized PA cost reporting at all stages, especially for MPAs and PAs in developing countries. The quantity and quality of such data presently limits research in PA sustainable finance, and will need to be remedied to advance the field in future years.

Keywords: conservation finance, protected area costs, protected area management, funding protected areas, sustainable finance, marine protected areas (MPAs), property rights

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****Correspondence:***
John J. Bohorquez
john.bohorquez@stonybrook.edu

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INTRODUCTION

Implementation of protected areas (PAs) for conservation restricts human activities, such as exploitation or extraction of natural resources, within targeted ecosystems. In so doing, PAs may preserve biodiversity in key areas, allow degraded ecosystems to recover, and increase resilience to the impacts of climate change (O'Leary et al., 2018). The rising popularity of PAs in recent decades is evidenced by the multiple global initiatives that have come into force to expand PAs

around the world (United Nations Environment Programme, 2011; United Nations Department of Economic and Social Affairs, 2017). These initiatives typically have separate goals for marine protected areas (MPAs) and land based PAs [which we refer to here as terrestrial protected areas (TPAs)]. Aichi target 11, formed in 2010 under the Convention for Biological Diversity (CBD), aims to have 10% of the ocean (within Exclusive Economic Zones) and 17% of land area protected by the year 2020. Following the CBD targets, the United Nations established a target of conserving 10% of the entire ocean by 2020 under Goal 14 Target 5 of the UN Sustainable Development Goals. IUCN has recommended an additional long term goal of protecting 30% of the ocean by 2030. In comparison, goals for TPA coverage have a particularly long history spanning multiple decades, including the IVth World Parks Congress under IUCN in 1992 that aimed to have 10% of each biome under protection by 2000 (IUCN, 1993).

In light of these PA expansion goals, there is much concern over how to generate adequate financial resources to achieve them. Costs that PAs incur are typically broken down into three categories (James et al., 2001; McCarthy et al., 2012; Brander et al., 2015), which are described as follows:

Establishment Costs – All costs in the time period from project conception up to active implementation. This may include purchase of land or other acquisition costs, administration costs, legal fees, transaction costs, research and surveys, and initial capital costs for enforcement equipment, tourism, or other capital infrastructure.

Operational Costs – year to year costs for management, monitoring, and enforcement. This may also include maintenance, scientific research for tracking PA performance, and employee salaries. Routine activities related to education and public or stakeholder outreach also fall under this category.

Opportunity Costs – the society wide benefits that are foregone by the restriction of economic activities resulting from implementation of the PA. Opportunity costs are frequently borne by external stakeholders, rather than the PA managing agency or institution directly. Opportunity costs may sometimes be a part of the budget for the PA in the event that management pays compensation to groups or individuals that lose income as a result of PA implementation.

A lack of adequate funding to meet PA costs both impedes the ability to expand PA networks, and may render existing PAs ineffective in reaching their conservation goals (Bruner et al., 2004; Edgar et al., 2014; Gill et al., 2017), regressing to what is referred to as "paper park" status (Thur, 2010). Furthermore, there is a general global funding gap for conservation at large. In 2014, Credit Suisse in partnership with McKinsey & Company, WWF, and Yale University released a comprehensive overview of the state of conservation finance. The report estimated that about $300-$400 Billion would be required per year to preserve healthy ecosystems around the globe, but only $52 Billion per year is actually being delivered (Huwyler et al., 2014).

While the funding gap applies to both marine and terrestrial conservation, there is a stark difference in the advancement of and available resources for TPAs versus MPAs. TPAs are much closer to reaching their expansion goal of 17% having achieved 14.8% coverage by the end of 2016 (Hussain et al., 2011; UNEP-WCMC and IUCN, 2016; United Nations Department of Economic and Social Affairs, 2017), whereas MPA coverage had only reached 5.1% by that time (UNEP-WCMC and IUCN, 2016). While MPA coverage has grown in recent years [estimated by the World Database on Protected Areas at 7.4% in October 2018 (Marine Protected Areas Coverage in 2018, 2018)], coverage still falls far short of the 10% by 2020 target, and fully or strongly protected MPAs comprise a small percentage of the total (Sala et al., 2018). In addition, marine ecosystems are particularly underfunded among PAs as a whole (Emerton et al., 2006; Bruner et al., 2008). Review of the currently available literature reveals that there are more studies of TPA costs than those of MPA costs, which only a handful of available sources directly address (Balmford et al., 2004; Gravestock et al., 2008; McCrea-Strub et al., 2011; Brander et al., 2015). The historic advancement of terrestrial versus marine conservation in implementation, research, and resource allocation fits with one researcher's description of marine ecosystems as the "Cinderella" of conservation (McIntyre, 1992 as quoted in Jones, 2014).

Financial mechanisms used to fund TPAs are currently more diverse and sophisticated than those used for MPAs. Thus, it may be instructive to consider successes in terrestrial conservation finance for MPA funding guidance (De Santo, 2012). Examples of traditionally TPA-oriented funding mechanisms that are now being applied to MPAs include debt-for-nature swaps (Gockel and Gray, 2011; Baird et al., 2017; Weary, 2017) and "blue carbon," or carbon offsets for coastal or marine specific sequestration (Murray et al., 2011; Pendleton et al., 2012; Runting et al., 2016; Baird et al., 2017). However, these mechanisms have only been employed in a handful of marine conservation projects around the world. In working toward achievement of the 10% by 2020 goal, MPA managers must continue to adapt TPA funding mechanisms and strategies to a marine context, and the growing emphasis on MPAs will present numerous opportunities for TPA finance experts to transfer their skillset to MPAs. However, for this collaboration and skills transfer to succeed, the fundamental differences between MPA and TPA finance need to be mutually understood.

This paper focuses on potential differences in costs between MPAs and TPAs, and how these relate to developing effective financial strategies. The aforementioned cost categories (establishment, operational, and opportunity) are differentiated based on whether costs are incurred upfront or whether they are ongoing year-to-year expenditures, which can make them more appropriate for some financial mechanisms over others. For example, revolving funds are used to provide immediate one time payments to recipients whereas trusts can deliver consistent payouts over a longer period of time (Clark, 2007). By definition, sustainable finance for PAs requires adherence not to just quantity of funds required, but also that funding is delivered in a timely manner in accordance with needs (Emerton et al., 2006). Thus determining any difference in the required timing of funds for MPAs versus TPAs is critical to adopting or designing effective and sustainable financial strategies. In this paper, we review the existing literature to investigate the potential for statistically analyzing the differences in the funding requirements

of MPAs versus TPAs. While we ultimately find that data are very limited, we provide some quantitative and qualitative insights, and provide recommendations as to data needs that will allow fuller elucidation of sustainable finance issues.

METHODS

Literature Review

We conducted a comprehensive literature review to examine costs incurred by the groups or agencies establishing and operating PAs. Per our focus on costs, we sought articles from all geopolitical scopes that provided specific values in currency terms, either estimated or actual observations, for any of the three cost categories (**Table 1**). We developed a collection of available literature via databases "Web of Science" and "Google Scholar" using combinations of keywords and phrases including; Marine Protected Areas, Protected Areas, Nature Reserves, Marine Reserves, Costs, Management, Operations, Expansion, and Establishment. We then expanded our collection by reviewing internal citations from this initial set of articles. We also considered articles uncovered by expert input, prior research, or conference attendance. Both peer reviewed and white paper or government reports were considered, as well as studies addressing multiple spatial scales, from site-specific assessments to comprehensive global estimates. The literature was collected in multiple stages; initially from September to November 2017, and then from January to February 2018.

Quantitative Comparison

Our intent is not to compare total funding requirements between MPAs and TPAs. Rather, we compare how total funding needs are distributed among the three categories of costs that PAs can incur during different stages of development. We framed this via a series of ratios that track proportional expenditures across cost categories:

Ratio 1: Establishment Costs as a % of Total Costs.
Ratio 2: Establishment to Operational Costs (EST : OP).
Ratio 3: Establishment to (Operational and Opportunity Costs) (EST : OP + OC).
Ratio 4: (Establishment and Opportunity Costs) to Operational Costs (EST + OC : OP).

Due to the differences between studies (e.g., scope, location, and time of study) that would influence costs, we did not combine different studies in our calculation of ratios. Rather, we calculated ratios from numbers provided within the same study to avoid confounding cost ratio results with study-related differences. Therefore, in our literature review, we paid special attention to studies that contained values for multiple cost categories as a prerequisite for inclusion in the quantitative comparison portion of the review.

Each study used a different time period for their respective analysis, with some incorporating discount rates when estimating future costs. For accurate comparison, we standardized to a common time period and discount rate based on the approach used in the Brander et al. (2015) report on future costs of global MPA expansion (Brander et al., 2015). Brander calculates establishment costs over a 5-year implementation period, immediately followed by 30 years of operations. Like Brander, we return present values (PV) for the 35-year period with a 3% discount rate. All dollar values are converted to 2017 USD via the Consumer Price Index (CPI) from the Bureau of Labor Statistics. Consequently, the methodology assumes establishment costs stretched over a 5-year period from 2018 to 2022, followed by 30 years of operational costs. Opportunity costs are factored in for the full 35-year time horizon.

RESULTS

Literature Review

Our literature review yielded twenty-four articles on PA costs for different spatial and political scales published from 1999 to 2018. We cataloged and present here (**Tables 2A,B**, **3**) each piece of literature based on scope, environment considered (marine or terrestrial), the costs reported, type of data, and eligibility for inclusion in our quantitative comparison. Papers and articles include projections for real cases, estimations for hypothetical scenarios, and observations from ongoing efforts Some studies are site- or region-specific, in which case their geographic focus is also referenced (Green et al., 2012; Rojas-Nazar et al., 2015; Pascal et al., 2018). Other studies have used a collection of case examples or data sets to construct cost models to both identify variables that influence costs, as well as to project costs of

TABLE 1 | Review of data type behind reported costs and geopolitical scope for surveyed literature in review.

Data types			
Reported observation Actual costs incurred by a single or set of protected areas that have been recorded and reported.	Surveyed estimation Costs projected for hypothetical expansion or implementation of new PAs via survey responses from PA managers or other experts.	Literature estimation Costs projected for hypothetical expansion or implementation of new PA via review of available literature.	Calculated estimation Costs projected for hypothetical expansion or implementation of new PAs calculated using existing model or other objective method.
Spatial/political scopes			
Site specific Cost numbers are attributed to a specific PA or local network of PAs.	National Aggregated costs for all PAs of a given type in a specific country.	Regional Aggregated costs for all PAs of a given type across a continent or region (e.g., Mediterranean Sea).	Global Aggregated costs for all PAs of a given type for a global target or hypothetical expansion scenario.

TABLE 2A | Literature review summary results for exclusively marine or joint marine-terrestrial studies.

Author	Year	Title	Peer reviewed?	Scale	Environment	Costs			Data type	Eligible for analysis?
						Establishment	Operational	Opportunity		
Klein et al., 2010	2010	Prioritizing land and sea conservation investments to protect coral reefs	Y	National (Indonesia)	Both		X	X	Estimations (calculated)	Yes
Gantioler et al., 2010	2010	Costs and socio-economic benefits associated with the Natura (2000) network	N	Regional (Europe)	Both	X	X		Estimations (surveyed)	Yes
Pascal et al., 2018	2018	Evidence of economic benefits for public investment in MPAs	Y	Site specific (Vanuatu and Saint Martin)	Marine	X	X		Observations	Yes
Binet et al., 2016	2015	Sustainable financing of Marine Protected Areas in the Mediterranean: a financial analysis	N	Regional (Mediterranean)	Marine	X	X		Observations and estimations (surveyed)	Yes
Brander et al., 2015	2015	The benefits to people of expanding Marine Protected Areas	N	Global	Marine	X	X	X	Estimations (calculated)	Yes
Rojas-Nazar et al., 2015	2015	Marine reserve establishment and on going management costs: a case study from New Zealand	Y	Site Specific (network of MPAs in New Zealand)	Marine	X	X	X	Estimations (calculated and surveyed) and observations	Yes

(Continued)

TABLE 2A | Continued

Author	Year	Title	Peer reviewed?	Scale	Environment	Costs			Data type	Eligible for analysis?
						Establishment	Operational	Opportunity		
United Nations Development Programme [UNDP], 2012	2012	Catalysing ocean finance volume I: transforming markets to restore and protect the global ocean	N	Global	Marine	X	X		Estimations (calculated)	Yes
Ban et al., 2011	2011	Promise and problems for estimating management costs of marine protected areas	Y	Site Specific (The Coral Sea, Australia)	Marine		X		Estimations (calculated and surveyed)	No
McCrea-Strub et al., 2011	2011	Understanding the cost of establishing marine protected areas	Y	Global	Marine	X			Observations	No
Gravestock et al., 2008	2008	The income requirements of marine protected areas	Y	Global	Marine		X		Observations	No
Balmford et al., 2004	2004	The worldwide costs of marine	Y	Global	Marine		X		Observations	No

Column for "Data Type" indicates whether costs were direct observation or estimations, and if estimations were developed by answers to surveys, calculated by the authors, or possible ranges via literature review. [Citations not included in text (Ban et al., 2011)].

TABLE 2B | Literature review summary results for exclusively terrestrial studies.

Author	Year	Title	Peer reviewed?	Scale	Environment	Establishment	Operational	Opportunity	Data type	Eligible for analysis?
Venter et al., 2014	2014	Targeting global Protected Area expansion for imperiled biodiversity	Y	Global	Terrestrial			X	Estimations (calculated)	No
Green et al., 2012	2012	Estimating management costs of protected areas: a novel approach from the Eastern Arc Mountains, Tanzania	Y	National (Tanzania)	Terrestrial		X		Observations and estimations (surveyed and calculated)	No
McCarthy et al., 2012	2012	Financial costs of meeting global biodiversity conservation targets: current spending and unmet needs	Y	Global	Terrestrial	X	X		Estimations (calculated)	Yes
Hussain et al., 2011	2011	The economics of ecosystems and biodiversity	N	Global	Terrestrial	X	X	X	Estimations (literature review)	No
Pearce, 2007	2007	Do we really care about biodiversity?	Y	Global	Terrestrial		X	X	Estimations (literature review)	No
Bruner et al., 2004	2004	Financial costs and shortfalls of managing and expanding Protected-Area systems in developing countries	Y	Global	Terrestrial	X	X		Estimations (literature review)	No
Moore et al., 2004.	2004	Integrating costs into conservation planning across Africa	Y	Regional (Africa)	Terrestrial		X		Estimations (calculated, using Balmford, 2003)	No
Balmford et al., 2003	2003	Global variation in terrestrial conservation costs, conservation benefits, and unmet conservation needs	Y	Global	Terrestrial		X		Observations	No

(Continued)

TABLE 2B | Continued

Author	Year	Title	Peer reviewed?	Scale	Environment	Establishment	Operational	Opportunity	Data type	Eligible for analysis?
Frazee et al., 2003	2003	Estimating the costs of conserving a biodiversity hotspot: case-study of the Cape Floristic Region, South Africa	Y	Site specific (South Africa)	Terrestrial	X	X		Estimations (calculated)	Yes
Shaffer et al., 2002	2002	Noah's options: initial cost estimates of a national system of habitat conservation areas in the United States	Y	National (United States)	Terrestrial	X	X		Estimations (calculated)	Yes
James et al., 2001	2001	Can we afford to conserve biodiversity	Y	Global	Terrestrial	X	X		Estimations (calculated)	Yes
TNC (Curtis et al., 2001)	2001	Planificacion financiera a largo plazo para parques y areas protegidas	N	Regional (Latin America)	Terrestrial		X		Estimations (calculated)	No
James et al., 1999	1999	A global review of Protected Area budgets and staff	N	Global	Terrestrial		X		Observations	No

Column for "Data Type" indicates whether costs were direct observation or estimations, and if estimations were developed by answers to surveys, calculated by the authors, or possible ranges via literature review (Citations not included in text James et al., 1999; Curtis et al., 2001; Shaffer et al., 2002; Frazee et al., 2003).

TABLE 3 | Number of articles by data type and geo-political scope.

Data types			
Reported observation	Surveyed estimation	Literature estimation	Calculated estimation
9	5	2	13
Spatial/political scopes			
Site specific	National	Regional	Global
4	3	4	13

*Some studies included multiple sources to generate their cost figures, e.g., both surveyed as well as calculated estimations (see **Table 2**).*

expansion on a global scale (Balmford et al., 2003, 2004; Moore et al., 2004; Gravestock et al., 2008; McCrea-Strub et al., 2011). These papers are particularly influential in the field as many other studies adopted their models for cost projections. Another subset of influential work includes papers that estimate costs for reaching specific global conservation goals, such as Aichi Target 11 and the UN's SDG 14.5 (James et al., 2001; McCarthy et al., 2012; United Nations Development Programme [UNDP], 2012; Brander et al., 2015).

Operational costs were the most commonly reported cost within the literature reviewed with 22 of 24 papers returning values. Establishment costs were less frequently reported, with 13 studies having figures. Only five studies returned values for opportunity costs, though this is partly attributed to our focus on costs being incurred by the PA, rather than costs borne by society at large. As mentioned in the introduction, opportunity costs would only translate to costs incurred by a PA managing agency if compensation is paid to those losing economic opportunities because of PA establishment. While there are examples of compensation packages for fishers as part of PA budgets like in the Great Barrier Reef (Macintosh et al., 2010), current literature suggests that such direct monetary compensation packages remain challenging especially for Marine PAs due to a lack of stakeholder use and activity data and are likely rare overall (McCay and Jones, 2011). Furthermore, it is possible that some of the studies we analyzed already accounted for such compensatory payments (and therefore a portion of opportunity costs) within their calculations for establishment and operational costs, perhaps contributing to the data limitations for opportunity costs. Hence, when discussing results going forward, operational and establishment costs are the primary focus and opportunity costs are a secondary consideration. However, we still accounted for and report ratios for opportunity costs when possible.

Quantitative Perspective

Fourteen works provided values across the required cost categories. Not all of these studies were eligible, however. Studies were deemed ineligible for the analysis if (1) costs were incompletely or inadequately reported or estimated (Venter et al., 2014), or (2) if they did not provide adequate distinction for how costs were distributed by category (Pearce, 2007; Hussain et al., 2011; Binet et al., 2016), marine vs. terrestrial environment, or a combination of both (Gantioler et al., 2010; Binet et al., 2016). Additionally, we removed Bruner et al. (2004) from eligibility as the only establishment cost estimate it provided was directly taken from James et al. (2001), which we already accounted for in the literature review (Bruner et al., 2004).

After filtering out these ineligible works, we were left with 10 studies to compare ratios against, only nine of which accounted for establishment costs. The combination of limited samples, combined with the fact that many samples were replicates of common estimation models (Balmford et al., 2003, 2004; McCrea-Strub et al., 2011), prevented us from conducting a detailed statistical analysis to test for a significant difference of cost ratios between MPAs and TPAs. Still, we provide an initial estimate of potential ranges and differences from the available data (**Figure 1** and **Table 4**). Most studies reported costs for a range of scenarios (e.g., cost estimates for total area protected versus cost estimation based on protection priorities (Brander et al., 2015), cost estimates based on MPA size (United Nations Development Programme [UNDP], 2012), cost estimates by wealth of country (McCarthy et al., 2012). The figures reported in **Figure 1** and **Table 4** correspond with the minimum and maximum ratios within each study across all scenarios presented.

Terrestrial protected areas generally hold higher values than MPAs for Ratio 1 (establishment costs as % of total) and Ratio 2 (establishment costs to operating costs). In all but one case, the exception being the minimum bound for Natura (2000), establishment costs for TPAs make up over 70% of total costs, and have a Ratio 2 of at least 2.68 implying that the majority of costs would be incurred prior to implementation. In contrast, establishment costs for MPAs make up a maximum of 39.40%, and all Ratio 2 values are well-below 1.00 such that the majority of costs are estimated to occur over the operational time period. In fact, there is no overlap in Ratio 2 values between MPAs and TPAs with the one exception again being the minimum bound of the Natura (2000) study, which slightly overlaps with the maximum values for MPAs under Brander and UNDP. Opportunity costs were only included in Klein et al. (2010), Brander, and Rojas-Nazar studies, so our ability to compare TPAs versus MPAs on the basis of Ratios 3 and 4 is quite limited. However, Ratio 4 is generally higher for the TPA than MPA scenarios.

DISCUSSION

Taking Stock of Current Literature

While our review is unique in its comparative focus on MPAs versus TPAs, it is not the first literature review to be conducted on costs incurred by PAs. Bruner et al. (2004) and Pearce (2007) are two examples included in our literature review that discuss a collection of literature assessing different types and methods

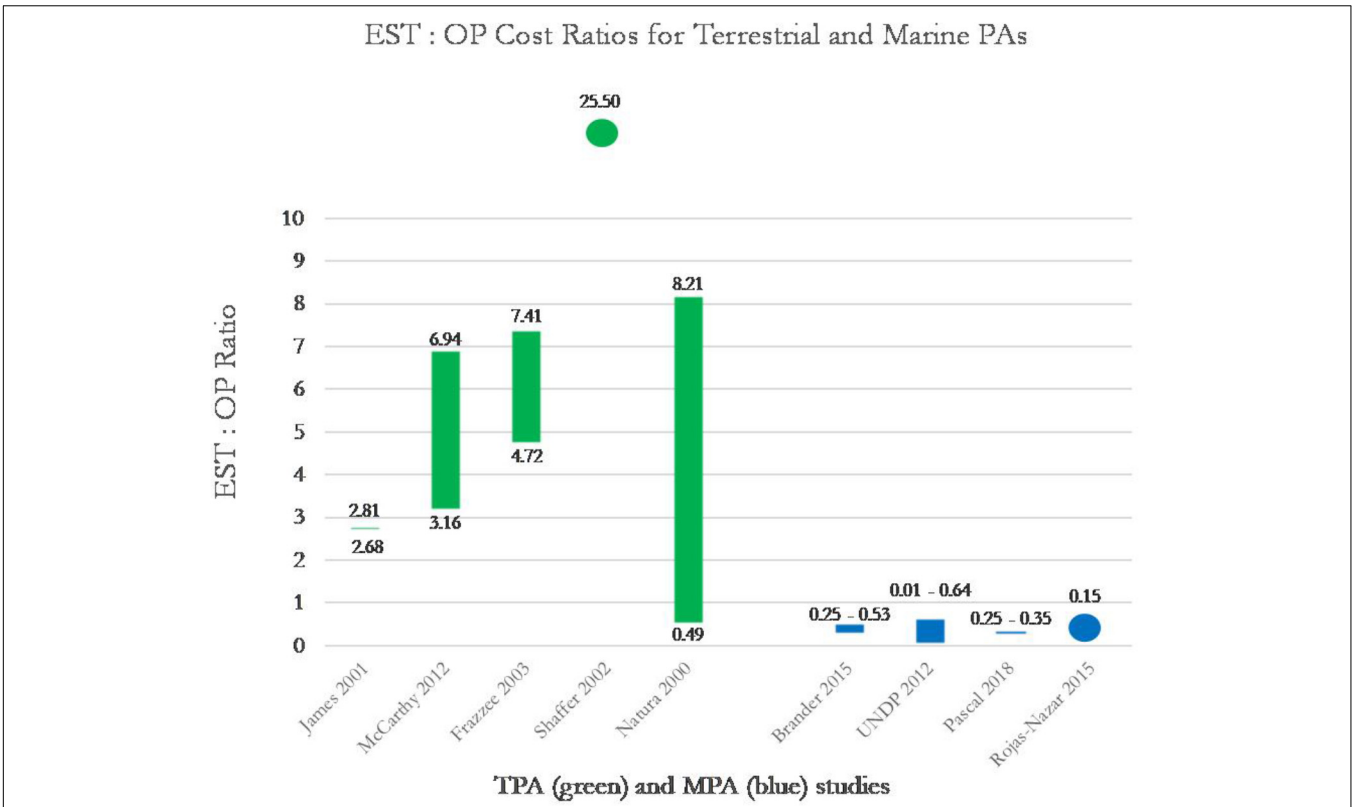


FIGURE 1 | Graphic demonstrating the range of Ratio 2 values (Establishment to Operational costs) calculated for five TPA and four MPA studies from **Table 4**. Bars indicate the boundaries of minimum and maximum ratio values for individual studies. Circles denote ratios for studies that only returned one value. The range of values are written next to their corresponding bars and circles.

TABLE 4 | Cost ratios per each eligible study.

	Ratio 1	Ratio 2	Ratio 3	Ratio 4
	EST as % of total	EST – OP	EST : OP + OC	EST + OC : OP
Terrestrial				
James et al., 2001	72.80 – 73.77%	2.68 – 2.81	2.68 – 2.81	2.68 – 2.81
McCarthy et al., 2012	75.97 – 87.41%	3.16 – 6.94	3.16 – 6.94	3.16 – 6.94
Klein et al., 2010*	N/A	N/A	N/A	1.30 – 13.99
Frazee et al., 2003	82.53 – 88.11%	4.72 – 7.41	4.72 – 7.41	4.72 – 7.41
Shaffer et al., 2002	96.23%	25.5	25.5	25.5
Gantioler et al., 2010**	32.91 – 89.14%	0.49 – 8.21	0.49 – 8.21	0.49 – 8.21
Marine				
Brander et al., 2015	2.63 – 12.52%	0.25 – 0.53	0.03 – 0.14	1.25 – 8.55
United Nations Development Programme [UNDP], 2012	1.16 – 39.40%	0.01 – 0.65	0.01 – 0.65	0.01 – 0.65
Pascal et al., 2018	8.71 – 20.10%	0.10 – 0.25	0.10 – 0.25	0.10 – 0.25
Rojas-Nazar et al., 2015	11.88%	0.15	0.13	0.23
Klein et al., 2010	N/A	N/A	N/A	0.002 – 3.78

**Klein et al., 2010 does not include establishment costs, so Ratio 4 is exclusively Opportunity Costs to Establishment Costs. **Scenarios included from Natura (2000) are for countries where MPAs were ruled out as the study did not adequately differentiate cost distribution by environment for countries where both MPAs and TPAs were present.*

calculating PA costs (Bruner et al., 2004; Pearce, 2007). However, both of these studies strictly look at terrestrial sources of funding. In addition, they both have similar faults and expose common gaps in information on this topic.

As discussed in the introduction, MPAs and financial research surrounding MPAs are generally not as in depth and widespread as their TPA counterparts. The focus of Bruner et al. (2004) and Pearce (2007) on terrestrial ecosystems is indicative of this trend.

Further, only nine studies focused exclusively on marine areas, with two additional studies looking at both types simultaneously. The materials included within Bruner and Pearce also indicate that establishment costs are much less frequently reported than operational costs for PAs in general. For example, while Bruner included a total of 15 studies, only James et al. (2001) included a direct estimate of establishment costs.

We came across several other informative sources during the research process that, while not included in our final tabulation, referenced the state of available information on PA costs. Several cited a lack of adequate data collection on conservation costs in general, including PAs (Naidoo et al., 2006; Ban and Klein, 2009; Kark et al., 2009; Cook et al., 2017). More specifically, data for marine planning and acquisition costs (as part of establishment costs) for PAs in developing countries have been cited as especially difficult to acquire (Balmford et al., 2003; Naidoo et al., 2006; Ban and Klein, 2009). This pattern is represented to a degree in our literature review with fewer works containing establishment cost estimates than operational cost estimates.

The dispersed nature of information within our review, as well as from qualitative references in other works, highlights specific gaps in the literature and directions for future focus in closing these knowledge gaps. Improving cost data in marine areas, and establishment cost data in both marine and terrestrial environments, needs to be a primary focus in order to improve assessment of financial sustainability for PAs. Researchers have also called for standardization of reported information in order to make data from different sources easier to compare (Binet et al., 2016; Cook et al., 2017). Such recommendations have included reporting of common line items or cost categories, as well as systematic methods of calculation and accounting. We experienced challenges ourselves from the lack of standardized reporting methodologies while trying to compare costs across different studies. Some experts and researchers have referenced global health programs as a bar for cost reporting that conservation efforts can try to emulate (Cook et al., 2017).

Quantitative Snapshot

Our comparison of the time distribution of costs obtained by tracking across multiple cost categories provides an initial understanding of differences between MPAs and TPAs that can be followed up by evaluation at an individual MPA and TPA level. We observe a common pattern where establishment costs make up a far greater share of costs for TPAs than MPAs. In the context of meeting financial needs, this may indicate that TPAs require a greater share of total funding requirements in costs leading up to implementation, whereas costs for MPAs are incurred on more of a long term year-to-year basis for management, monitoring, and enforcement.

While we are limited in observations pertaining to opportunity costs, Klein et al. (2010) estimate higher opportunity costs as a proportion of management costs for terrestrial regions than marine areas. In that study, opportunity costs were influenced by agricultural rents and income from fishing for TPAs and MPAs respectively, indicating that compensatory payments could perhaps be higher for infringement on land development than extraction of marine natural resources. However, the nature of and amount to which these opportunity costs might result in compensatory payments is likely variable across countries. For that reason, it is important to have a globally representative spread of PA cost data to get a complete picture of PA costs, rather than, for example, data from strictly developed countries that may have better reporting capacity.

Theoretical Interpretation

Below we provide potential theoretical support for the observed findings on differences in cost ratios in TPAs versus MPAs. Similar to our quantitative comparison, the intent is to provide an initial perspective on TPA versus MPA costs and their implications for financing strategies.

Property Rights

Perhaps the greatest fundamental difference between MPAs and TPAs is the prevalence of private property rights in policies and spatial management of land versus sea. Purchasing private property rights as an establishment cost should therefore theoretically play a greater role in the costs of TPAs than MPAs. Private property rights are generally more prevalent in land-based scenarios due in part to the relative ease of identifying and establishing boundaries (Jones, 2014). Parties that can hold private property rights include individuals (for residence and commercial use), corporations, and in some cases communities that restrict use of land to community members. Some of the most frequent commercial uses for private land include agriculture or timber, which according to Maxwell et al. (2016) are also the two greatest threats to terrestrial biodiversity at large (Maxwell et al., 2016). The widespread threat of agriculture to terrestrial conservation, and the frequency with which TPAs are likely to require purchases of private land, is also demonstrated by many studies in our review that incorporate agricultural land values into considerations for estimating costs. Such examples include the McCarthy et al. (2012) study that exclusively relied on agricultural land values to estimate global costs of TPA expansion. In addition, Klein et al. (2010) and Venter et al. (2014) studies incorporate opportunity costs as a function of agricultural rents, and James et al. (2001) and Shaffer et al. (2002) used land market values to calculate purchase prices under establishment costs. Despite slight differences in methodologies, we find that TPA studies commonly calculate expansion costs as directly related to the value of private property and use rights, and thereby view expansions as directly imposing onto private land in the majority of cases.

For the ocean, quasi-property rights can be introduced spatially via mechanisms such as Territorial Use Rights for Fishing (TURF) that give permitted vessels exclusive access over certain fishing grounds. In addition to zonal rights, access rights can be allocated to specific uses across a marine area, including to specific resources and industries. Not only are private property rights very rare in ocean regions (Jones, 2014), but the ocean and its resources have also been generally viewed through the lens of open access, such that development of private property rights is frequently considered a form of conservation itself. In some cases, areas with private property rights for marine resources have been considered de facto MPAs, also referred to by state

agencies as DFMPAs (National Marine Protected Areas Center, 2008; Jones, 2014). One example is the leasing of marine areas for offshore wind energy, which has been discussed as having positive conservation benefits for the restrictions placed on fishing in such areas (Coates et al., 2016; Hammar et al., 2016).

Therefore, the use of private property rights as a conservation measure suggests that future MPA expansions may be less likely to encroach on regions where marine private property rights presently exist, whereas TPAs are likely to target areas with private property rights to restrict industries like timber, mining, and agriculture that are substantial threats to terrestrial conservation. Property rights (or the lack thereof) pose an interesting paradox for marine conservation in that a lack of property rights has frequently been associated with over exploitation and ecosystem degradation, yet may also provide an opportunity to establish MPAs at a lower cost than if property rights were more widespread.

Logistics and Operational Costs for Management

When analyzing costs of PAs, it is important to consider the logistical differences between required management, monitoring, and enforcement activities in marine versus terrestrial environments. While such differences have never been directly compared in a quantitative manner, studies have outlined general differences between marine and terrestrial conservation. In one example of a feasibility assessment for MPAs in Sweden, the authors cited Swedish administrative officials as claiming that MPAs are substantially more expensive to manage than TPAs, including monitoring and enforcement (Grip and Blomqvist, 2018). This was primarily attributed to a need for ships and advanced technology required for monitoring and enforcement in a marine environment. While ships are expensive to purchase, the operating costs of vessels is also particularly high. For reference, a recent study on MPA monitoring assumed $30,000/day for ship time (Kachelriess et al., 2014). While this estimate is specific to larger offshore vessels, even the smallest vessels for coastal or nearshore monitoring can cost 100s of dollars per day.

It is also important to consider potential differences in PA size between marine and TPAs and their influence on logistics and thereby operational costs, especially because comparative research finds that MPAs are larger on average than TPAs (Lindholm and Barr, 2001) and that the largest PAs in the world are typically MPAs[1]. Larger PAs generally require higher total operational costs. However, research indicates that larger PAs have lower operational costs per unit area than smaller PAs (Balmford et al., 2003, 2004). Thus, in the case of our calculations for area based changes in the United Nations Development Programme [UNDP] (2012) report, larger MPAs return a higher Ratio 1 value (EST : OP) than smaller MPAs despite having higher operational costs overall (see **Supplementary Material**), presumably as the decrease in marginal establishment costs per unit area is not as substantial.

[1] www.protectedplanet.net

There has been much recent focus on the prospects for technological advances to lower costs (Grip and Blomqvist, 2018). Remote monitoring is an increasingly popular method for marine and terrestrial ecosystem surveillance that can reduce the need for active vessels and vehicles, with subsequent promise for cost savings and improved execution for enforcement and monitoring (Pala, 2015; Proud et al., 2016; Richards et al., 2017). For enforcement purposes, improvements in remote monitoring may benefit MPAs more than TPAs considering that land based poachers and other violators can hide under forest canopy and other terrestrial features. For monitoring ecological performance, remote sensing remains limited to surface layers of the ocean, and expensive (and sometimes environmentally harmful) *in situ* monitoring tasks such as SCUBA diving and benthic trawls are often needed (Pomeroy et al., 2004). However, further improvements in remote monitoring and advancements in other cheaper and less invasive *in situ* methods such as environmental DNA, drones, satellite images, etc. may lead to significant cuts in operational costs required for MPAs in the future (Bohmann et al., 2014; Pikitch, 2018). Future research should evaluate how the addition of new technologies may benefit MPAs and TPAs differently depending on the specific technology and PA context.

CONCLUSION

The results of our literature review revealed a lack of available data to statistically analyze differences among three categories of costs incurred by PAs. Still, our findings provide an initial perspective on how MPAs and TPAs may incur costs differently. We observe a distinct pattern in the presently available information where TPAs incur a greater proportion of costs prior to implementation, while MPAs typically incur the majority of costs over the long term. Per our observations, TPAs would ideally focus on financial strategies that can deliver the majority of total required funding prior to implementation. Meanwhile MPAs may be better candidates for strategies that can guarantee consistent and controlled funding over multiple decades. While such a pattern between one-off implementation costs and ongoing costs seems elementary in theory, perhaps it has not been given proper recognition in practice as evidenced by a lack of financial resources made available to long term MPA operations and resulting paper park status (Reid-Grant and Bhat, 2009; Thur, 2010; Gill et al., 2017). And if MPA managers have yet to give this due consideration, then this lesson is all the more relevant for terrestrial conservation finance professionals looking to focus on the many growing opportunities in marine conservation. One recent example of successfully adapting a traditional TPA funding metric to an MPA is the debt-for-nature swap in the Seychelles orchestrated by The Nature Conservancy, which includes a regimented funding plan for at least 20 years of marine conservation efforts in the country (Debt Relief for dolphins: A new plan to protect the water around the Seychelles, 2017).

Our study also leads to an even more important conclusion about research surrounding PA costs and finance. This review documents that presently available PA cost data and statistics

are insufficient to answer basic questions about PA costs and funding needs on a technical level. The inability to rely on rigorously collected data to conduct specific analyses will likely limit advancements in PA sustainable finance until the data gap is remedied. Furthermore, in addition to the need to expand the amount of information available, it is necessary to transition to a network of higher quality data. Only eight of the 22 studies in our literature review included any actual observations, whereas the majority of cost assessments were estimations determined either by a method of calculation or response to a survey, including all but one of the studies we were able to adapt to our quantitative comparison. To refine our understanding on PA costs and management, we need to transition from data rooted in estimations projecting hypothetical scenarios toward actual PA observations. More investment is therefore needed in both MPA and TPA conservation cost reporting, ideally in a standardized metric as recommended by other researchers in the field (Cook et al., 2017).

AUTHOR CONTRIBUTIONS

JB compiled and synthesized the background research for this work, including the literature review. He also structured and performed the comparative analysis, and organized and wrote the initial drafts of the manuscript. AD provided guidance on the development of the paper and its structure, reviewed the quantitative findings and suggested modifications, edited versions of the manuscript, and assisted in conceptualizing the implications of the research. EP contributed to the conceptualization, writing, broader context, and provided intellectual insight to the manuscript. She also helped obtain the funding with which this work was supported.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00045/full#supplementary-material

TABLE S1 | Costs and calculated ratios for individual conservation scenarios within each study assessed in the quantitative comparison.

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Introducing Relational Values as a Tool for Shark Conservation, Science, and Management

Rachel A. Skubel, Meryl Shriver-Rice and Gina M. Maranto*

Abess Center for Ecosystem Science and Policy, University of Miami, Coral Gables, FL, United States

Relational values (RV) are values that arise from a relationship with nature, encompassing a sense of place, feelings of well-being (mental and physical health), and cultural, community, or personal identities. With sharks, such values are formed by diverse groups that interact with these animals and their ecosystems, either physically or virtually, whether a scientist, student, fisher, or media-viewer. Further, these user groups may overlap or come into conflict over management plans, media portrayals of sharks, and their conservation status. Although scientists have not explicitly aimed to assess RV through sharks, qualitative studies of shark fishers, tourism operators, tourists, and the public, as well as historical and archeological accounts, can be interpreted through an analytical lens to reveal values which can also be defined as relational. To this end, this review considers studies capturing RV alongside those of economic value (increasingly, the value of a shark is appraised by their financial value in shark tourism) and the social and cultural roles of sharks. Based on these studies and the broader RV literature, we then outline a workflow for how RV can be leveraged in scientific inquiry, equitable resource management, and education. We conclude that via collaborative assessments of RV, with implicit inclusion of multiple values of sharks and by acknowledging their importance to all parties involved in user conflicts, the RV framework can lead to a constructive dialog on polarizing conservation and management issues. By illuminating shared values, and/or revealing dichotomies of values ascribed toward certain areas or objects, this framework can provide inroads to mediation, seeking to conserve or even restore relationships with nature, and their derived values as much as is possible. This approach can yield unexpected knowledge, solutions, and compromises in an increasingly complex conservation landscape.

Keywords: relational values, ecosystem services, well-being, sharks, elasmobranchs, conservation, management, values assessment

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****Correspondence:***
Rachel A. Skubel
rskubel@gmail.com

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INTRODUCTION

Why Value Sharks?

The human-nature relationship has been understood in a myriad of ways by various cultural and ethnic groups through time. Plants, animals, and their ecosystems are acknowledged to play significant ecological roles, while also playing a role in human societies – this latter role is more difficult to quantify and has typically been approached through an "ecosystem services" framing. Finding a way to intercompare these diverse

services from nature, from well-being and cultural importance to direct financial gain (e.g., from exploitation for industry) (Turner et al., 2003), is important for inclusive decision making and policy. For instance, in planning for development or conservation, such as through impact assessments and choosing among alternative courses of action (Nelson et al., 2009), and in addressing environmental degradation through equitable remediation, loss compensation, and resource allocation (Bladon et al., 2018). Throughout this paper, we refer to either values, ecosystem services, or both, depending on the context at hand. These terminologies are, in some respects, interchangeable in referring to human benefits from nature, but in certain policy-specific scenarios, it becomes important to use whichever term is collectively agreed upon by practitioners. We have elected to focus on the emerging framing of "relational values (RV)," or values from the human-nature relationship. This framing, which has gained momentum in academic and policy circles since its introduction in 2015 (Chan et al., 2016; Pascual et al., 2017), aims to improve upon prior value classifications which distinguished nature as something to be valued by humans, and in doing so, set humankind apart from what was thought of as "nature." Given the intractability of present and future human society from the world's ecosystems, the RV approach is a promising framework through which to interpret modern environmental problems which impact humans and non-humans alike.

Sharks and their relatives are a particularly threatened natural resource: of the 1041 species of sharks, skates, rays, and chimeras, one quarter are threatened by overfishing (Dulvy et al., 2014). In this paper, "sharks" is used to refer collectively to Chondrichthyan species – that is, sharks, skates, rays, and chimeras. Policy-makers and resource managers encounter many compounding barriers in attempting to conserve shark populations, spanning across aspects from ecological and biological to institutional, economical, and sociological (Chin et al., 2010; Bornatowski et al., 2014; Dulvy et al., 2017; Jabado, 2018). Although some "bright spots" of shark fishery management have emerged (Simpfendorfer and Dulvy, 2017), many populations are at risk of extinction due to historical over-exploitation, and a life-history pattern which lends to relatively slow recovery (e.g., larger sharks live relatively long, and reproduce small litters on a several-yearly basis) (Compagno, 1990; Field et al., 2010). Globally, 40% of shark catch originates from seven countries with low human development indices (Indonesia, India, Pakistan, Yemen, Tasmania, Nigeria, and Senegal), where sharks can be a source of both income and protein (Dulvy et al., 2017). Of these countries, a review of shark and ray conservation priorities found that India and Indonesia were some of the least likely countries to take conservation action (via fisheries management), despite hosting a high number of endangered and critically endangered species (as per The International Union for the Conservation of Nature (IUCN) Red List of Threatened Species) (Dulvy et al., 2017). Implementing realistic, achievable, and sustainable conservation measures for sharks call for not only an investment of financial resources to monitor the shark populations in question, but also a significant investment of human capital to assess values derived by those individuals and communities who are implicated in conservation plans.

What Are Relational Values?

The RV framing is inclusive of all values that can arise out of a person's or society's relationship with nature (Chan et al., 2016). Past value frameworks, such as the "cultural values" section of the millennium ecosystem assessment (MEA), encountered difficulties because these values were not well defined, and challenging (even counterproductive) to conceptualize for monitoring or measurement, as part of a quantification-focused model (Kenter et al., 2011, 2015). RVs may help to facilitate a dialog over where or whether to draw the human-nature divide in the assignment of value, by embracing the intractability of human society and natural systems. Although studies explicitly aiming to assess RVs are as of yet few, such values have been assessed in other disciplines, or used different terms. Environmental values (Kempton et al., 1996; Satterfield, 2001; Schneller et al., 2013), emotional attachments (Nightingale, 2012), morals and values (Colding and Folke, 2001; Peterson et al., 2002; Daw et al., 2015), social and community identity (Stoffle et al., 2009; Mccright and Dunlap, 2015), and stewardship/conservation ethic or awareness (Lucy and Davy, 2000; Lynch et al., 2010; Whatmough et al., 2011; Shiffman and Hammerschlag, 2014) have all been assessed with either quantitative (e.g., economic assessments by academic researchers or non-governmental organizations, ecological indicators) or qualitative (e.g., sociological and anthropological methods such as interviews and participant observation, by academic researchers or conservation practitioners) means. Other studies have taken a broader view of "ecosystem services" (ES) which include relational, intrinsic, and instrumental values (Hicks and Cinner, 2014; Fischer and Eastwood, 2016; Lau et al., 2018).

This review will summarize and contextualize the nascent field of RV, with a focus on human interactions with elasmobranchs (sharks, skates, and rays), and how their value to society has been assessed and discussed. Given the predominance of economic valuations in the marine conservation literature, we will briefly outline the applications, and issues associated with this approach, contrast economic valuations with more holistic and mechanistic approaches, and introduce a "multi-pronged" approach to valuation of elasmobranchs which includes RVs. Our approach acknowledges the perceived ecological, spiritual, cultural, financial, academic, and recreational significance of elasmobranchs in human experience, and envisions how considering such pluralistic values when formulating conservation strategies or interventions might yield outcomes that are more equitable and effective for human and non-humans alike.

METHODOLOGICAL APPROACH

In our search of the literature, we defined RV as values that arise from a relationship with sharks (studying, fishing, and observing), encompassing sense of place, feelings of well-being (mental and physical health), and cultural, community, or

personal identities (Chan et al., 2016; Pascual et al., 2017). We took the view that such values were formed by different groups that interact with sharks, physically or virtually – whether scientist, student, fisher, or media viewer. Although scientists have not explicitly aimed to assess RV through sharks, qualitative studies of shark fishers, tourism operators, tourists, and the public, as well as historical and archeological accounts, can be interpreted through an RV lens to reveal values which can also be defined as relational as per Chan et al. (2016). To this end, we included studies capturing RVs, alongside those on economic value (increasingly, sharks' value is appraised by their financial value through shark tourism) and included sources to consider how RV can be leveraged in scientific inquiry, resource management, and education.

To guide interpretation of the literature, we addressed the following set of questions:

(1) How are RVs formed?
(2) How are values formed with respect to sharks?
(3) Do different shark "user groups" overlap or conflict in their "use" of sharks, and/or their perceptions of the value of sharks?
(4) How can RVs be harnessed as a tool for shark research and conservation?
(5) How can researchers and conservation/management practitioners conduct assessments that incorporate the RVs of sharks?

RESULTS AND DISCUSSION

How Are Values Formed?

Valuations of services or benefits of natural resources is common practice in the policy and management space (Turner et al., 2003; Clifton et al., 2014; Luisetti et al., 2014; Arkema et al., 2015); however, recognizing that a value in itself might be meaningless, some researchers have applied post-hoc theoretical frameworks to explain either the underlying "drivers" of such elicited values, or how those values were formed by a person or group of people. The most relevant such framework may be the theory of "Emotional Affinity" with nature, introduced by Kals et al. (1999) which offers a conceptual framework for situating RVs and values toward nature in general. This notion is built on the notions of emotional bonds and ties with nature, which drives a person's desire to protect it (Vining, 1992; DiEnno and Thompson, 2013), and is couched not only in the importance of spending time in nature, but on sharing such positive experiences and feelings in nature with others (Kals et al., 1999; Curtin, 2005; Edwards et al., 2016). This is similar to the "biophilia" hypothesis that because their well-being is dependent upon it, human beings are inherently attuned to nature and this drives their desire to protect it (Kahn, 1997; Robinson, 2001). Collectively, these emotional and social ties around nature can heighten the perceived psychological cost of not acting to protect nature, even when one person's actions may seem insufficient in scale to solve the problems at hand. This framework provides a lens through which to interpret values from sharks, whether values are explicitly defined as "relational," or if situations are described in which RVs might emerge.

Another approach is the Value-Belief-Norm theory which originated from Stern et al. (1995) and Stern (2000). Value-Belief-Norm theory postulates a causal relationship among five variables which leads to a behavior: "values" (biocentric, altruistic, and egoistic) forming the base of this causal chain, leading to a set of beliefs – an ecological worldview, enabling the perception of risk to "valued objects," and a realization of one's own agency to reduce threats to that object of value. These beliefs, arising from the initial set of values, then precipitate a "sense of obligation to take pro-environmental actions," which manifest as behaviors such as activism, non-activist public sphere or private sphere behaviors, and behaviors within an organization (e.g., as part of an environmental group) (Stern, 2000). **Figure 1** shows how the Emotional Affinity and Value-Belief-Norm theory explain pro-environmental actions from similar yet distinct theoretical approaches.

How Are Values Formed With Respect to Sharks?

Cultural Identities: Sharks as a Symbol and a Resource

Indigenous relationships with the sea

Archeological records and oral histories of pre-colonial societies have shown sharks' dual role as a fishery resource and a cultural symbol related to personal and community identities, foreshadowing shared RV with modern shark fishers. Polynesian Māori arrived in New Zealand ∼ 1280 A.D. (Wilmshurst et al., 2008), where they practiced self-regulated shark fishing, enforcing penalties for fishing outside of permitted times. Sharks are represented in Māori folklore and oral history as both a dangerous predator and an example of strength to aspire to, as a symbol in artwork, and as a source of materials for tools and weapons. Similarly, archeological records from the Americas show a long (thousands of years) history of the use of sharks. Archeological sites of pre-Columbian indigenous societies, located in present-day Florida, yield shark teeth tools and weapons (cudgels), showing their instrumental value, and the wide presence of elasmobranch centra (vertebrae) in Floridan sites [e.g., 14,000 centra across 64 of 96 sites examined by Kozuch (1991)] emphasize the role of sharks as a multifaceted resource (Wing and Loucks, 1982; Walker, 2000). Through investigation of Mayan and Mexican sites, researchers have suggested shark teeth were used ceremonially (De Borhegyi, 1961), as well as for their instrumental value as food – shark liver oil being particularly high in vitamin A and D. Sharks may have been a highly valued nutritional resource in these societies (Kozuch and Fitzgerald, 1989). For Pacific societies such as the Maori or in the Gilbert Islands, ethnohistorical studies from the 1900s were able to describe fishing practices and the cultural role of sharks through interviews (Luomala, 1980). However, there were too few remaining members of indigenous groups in the Americas to provide these accounts, so hypotheses on the value of sharks must be based on physical evidence alone.

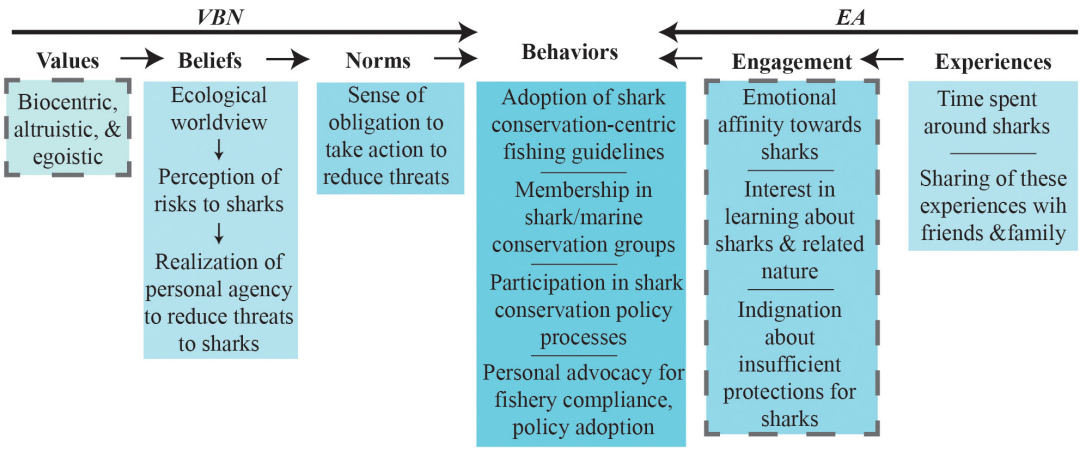


FIGURE 1 | Values have been explained in the context of theoretic frameworks leading to pro-environmental behaviors; here, we show how the Value-Belief-Norm and Emotional Affinity frameworks propose mechanisms leading to a behavior, in the context of sharks. Dashed lines indicate the component of the framework most similar/relevant to "relational values." This figure is based off of schematics from Stern (2000) and Kals et al. (1999).

In present-day New Zealand, Jøn and Aich argue that white sharks are perceived as at odds with fishing and beach tourism, yet beneficial for shark-associated tourism (Jøn and Aich, 2015). In French Polynesia, where sharks were also associated with warrior identities, fishing for sharks was a ritualized practice, including species-specific methods of fishing, and in some areas, self-enforced limits on catch (e.g., catching certain sharks or fishing in certain areas or times was seen as "taboo" given sacred associations with sharks, places, or times) (Torrente et al., 2018). Sharks appear in the identities and oral traditions of present-day "Saltwater People," a subset of indigenous and native islanders in northern Australia whose livelihoods and spirituality are dependent upon the marine environment (McNiven, 2004). Here, the Rrumburryia clan of the Yanyuwa people tells the story of "The Tiger Shark (Ngurdrungurdu) Dreaming," which conveys the journey of a shark throughout the northern region of the continent, interacting with humans and other land-bound animals along its way (The Yanuwa People, 2008). This story exemplifies how sharks are woven into the human-nature relationship, to the extent that the tiger shark in this story can represent an extension of the storyteller's identity in its retelling.

Eating sharks: What is the connection between sustenance and stewardship?

We focus this discussion largely on Hawaii and China, due to their predominance in the literature regarding historical culinary utilization of sharks, and because comparing these two regions demonstrates how specific ways of relating to sharks, enabled by geography and cultural trajectories, have persisted from prehistorical periods to the present day. In Hawaii, archeological evidence indicates that before European colonial contact, sharks were considered a "luxury" or elite food item, the consumption of which was synonymous with elevated high social status (Kirch and O'Day, 2003). In China, shark fin enjoyed a similar culinary status, as early as the Sung dynasty (AD 960–1279). A comparison of these two cultures suggests that the Chinese consumed sharks because of (a) a belief that the consumer would be imbued with the strength of the shark by association, and (b) shark fin (consumed in soup at present) was associated with wealth and prestige; whereas, in Hawaii (similar to Mâori and other Pacific-associated cultures), the shark held both a mythical, cosmological, and spiritual significance (Mokuau and Browne, 1994; Dell'Apa et al., 2014). Interpreting these different meanings behind shark consumption through a relational value lens, however, shows that all three traditional societies drew meaning and identity through their interaction (fishing, consumption) with sharks, versus utilitarian nourishment alone. Over time, however, the scale of consumption reached higher levels in China than either New Zealand or Hawaii, due to population growth as well as the fact that shark fin continues to be a symbol of wealth, status, and strength, within a trade infrastructure where sharks and shark fins are an economically viable product (Dent and Clarke, 2015). The Hong Kong SAR of China accounted for 44–59% of global shark fin imports in 2000, and although this proportion has since declined, the region is ranked as the top global trader of shark fins for 2000–2011 (Clarke, 2004; Dent and Clarke, 2015; Grimes, 2018). There are two further distinctions in valuation among these three regions: firstly, in China, consumption of sharks contributes to personal identity, whereas in Hawaii, sharks were both a "family god," *aumakua*, as well as being a "personal god," *akua* (Mokuau and Browne, 1994). Secondly, both New Zealand and Hawaii are more maritime oriented regions; while native Hawaiians regularly saw sharks in the wild, associating these interactions with mythical and spiritual meaning (Mokuau and Browne, 1994), and similarly, the Maori of New Zealand fished for sharks as part of a tradition linked with folklore and culture (Jøn and Aich, 2015), most Chinese consumers do not experience the same interaction and association with nature when consuming sharks.

China provides another example of distancing the act of consumption from the action and experience of catching; as the world's top producer of wild seafood, more of this catch

comes from outside China's waters than within (Mallory, 2013). Thus, RV in China toward sharks might not have relevance to a nature conservation or sustainability framing unless consumers and traders can form this association through targeted education and outreach (Tsoi et al., 2016). Underscoring the importance of establishing these links, a recent genetic study on the origin of shark fins in the Hong Kong retail market by Fields et al. (2018) found that less than 10 of 76 species identified were associated with sustainably managed fisheries, and approximately one third of species were at risk of extinction as defined by the IUCN Red List of Threatened Species. One challenge, however, of conservation initiatives focusing on shark consumption, is that many consumers may be unaware they are eating a shark species – whether they do not recognize the name of the product as indicating a shark (e.g., 77.5% of survey respondents who had eaten "tollo" meat did not know they had eaten shark meat, which this name refers to), or, a seafood product can be mislabeled, so that a consumer inadvertently buys shark meat for consumption (Jacquet and Pauly, 2008).

Value Formation Through Physical Interaction: Tourism and Fishing

Fishing for sharks: A source of values and conflicts

The practice of fishing can contribute to mental well-being through self-actualization (challenge, adventure, and independence), to the extent that some fishers will not choose alternative livelihoods offered by government incentives aiming to reduce fishing pressure (Pollnac et al., 2006, 2015). By fishing for a living, or as a pastime, individuals can form personal identities which include the ocean (e.g., see themselves as "a part of the sea," or the sea as "a part of them") (Nightingale, 2012; Voyer et al., 2015). This identity is echoed in studies of recreational shark fishers and operators (Barrowclift et al., 2017). Notably, many fishers saw their own knowledge and perceptions of shark population trends as being at odds with regulations, and by extension those regulations as an unmerited challenge to their ability to fish (Lynch et al., 2010; Shiffman and Hammerschlag, 2014; Shiffman et al., 2017). Studies of recreational shark fishers in Florida and Australia found that fishers generally displayed pro-conservation behaviors and attitudes; for instance, most were aware that "catch-and-release" fishing, where the shark is released instead of taken, serves to maintain the functional role of sharks in their environment (Lynch et al., 2010; Shiffman and Hammerschlag, 2014; Heard et al., 2016). In some cases, fishers had some understanding that certain shark species were less likely to survive after catch and release fishing, and tried to release sharks in a good condition to improve their chances of survival (Lynch et al., 2010), while in others, these attitudes were not accompanied by fishing gear to improve shark survival (Heard et al., 2016). In many cases, people fishing for sharks as part of their livelihoods (e.g., commercial or industrial fishing, depending on the terminology used, or recreational fishing businesses), are not doing so as part of a "target" shark fishery; instead, they are seeking other species, such as tuna, but also catch sharks, which they might then turn into a commodity (e.g., if there is a market or dealer to whom they can sell shark meat or fins, whether or not it is legal to do so) (Jabado et al., 2015). Recreational fishing businesses may market several different species-focused expeditions, one of which may be sharks (Shiffman and Hammerschlag, 2014). Indonesia represents one of the few directed/targeted shark fisheries; where sharks play a significant role in Indonesian fishing communities, there is greater economic significance from the fluctuations of local shark populations (Jaiteh et al., 2016a,b, 2017). From a values assessment perspective, to these communities, sharks' economic value may be of great importance, whereas in other communities where sharks are of less relative financial importance versus other fishery species, other values from the shark fishing activity, such as RVs, might be more apparent than the economic value.

In regions where fisheries monitoring data is limited or lacking, fishers can be a source of knowledge for long-term population trends; this knowledge has been recognized by researchers through the surveys of traditional or local ecological knowledge (TEK or LEK). In the Gulf Region of the United Arab Emirates, artisanal or industrial fishers have for decades been catching sharks for some part of their income, which was leveraged by Jabado et al. (2015) through LEK surveys to establish baseline information on the abundance and sizes of sharks. Similar to other long-term users of marine areas (Suman et al., 1999; Gray et al., 2010; Nayak, 2017), participating fishers felt that their knowledge was not sufficiently consulted during management planning, which in turn affected their ability to access fish (sharks and otherwise).

Given that fishing is important for sustaining their livelihoods, way of life, and well-being (all of which entail RV through the act of fishing) these individuals are motivated allies of conservation and planning. Complementary to the positivistic nature of most academic fisheries, knowledge provided by fishers is "interdisciplinary" by nature, as they make decisions - such as where to fish – based upon a combination of meteorological, oceanographic, biological, and social information (Thorlindsson, 1994). When designing management plans or performing values/ES assessments, one way of implicitly including RV can be by considering what "success" means to a fisher (e.g., a good catch, having sources of information on where to fish), versus a scientist (e.g., publications, scientific discovery) (Thorlindsson, 1994). However, the value of this partnership may go unacknowledged by fishery managers, presenting an untapped resource for gathering ecological data, while fostering stewardship through acknowledging the expertise of fishers. In developing nations, researchers associated with academic institutions are gathering experiential knowledge such as shark abundance, size trends, and market values through questionnaires, interviews, or by employing community members as data collectors (Jabado et al., 2015, Jabado, 2018; Jaiteh et al., 2016a; Humber et al., 2017). These findings are shared with the academic community and others with access to scientific journals, however, pathways on how to use this in management are not explicit. There is an emerging awareness of the importance of knowledge brokering with decision makers in order to reach conservation goals (Cvitanovic et al., 2016); accordingly, scientists might consider their role in fostering knowledge exchange as feasible when gathering experiential data. Researchers have served this role in using local knowledge

to initiate management action, although social, economic, regulatory, and cultural constraints can dampen efforts; in a successful instance, Heyman et al. (2001) used an extensive visual survey to investigate reports of a whale shark aggregation in the Gladden Spit area of Belize, an aggregation associated with important spawning events of reef fish. These results led to the establishment of a marine protected area to protect the spawning fish (Drew, 2005).

Diving with sharks: Harnessing emotion and social interaction for value formation

There is preliminary evidence from Apps et al. that stewardship values could be fostered through shark diving tourism: here, cage diving with white sharks was associated with an increased desire to contribute to shark conservation efforts, particularly when strong emotions were experienced during the shark-human interaction (Apps et al., 2018). However, another study of shark diving tourism participants found that although knowledge of sharks increased, pro-environmental attitudes did not shift from pre- to post- dive (Smith et al., 2014). Both cases suggest that short-term tourism experiences featuring shark interaction (e.g., over the course of 1 day) might increase knowledge or awareness of sharks and associated conservation challenges, particularly if the diving experience is paired with teaching or a presentation by guides (see also Apps et al., 2017). However, attribution of value formation to these experiences is unclear: expansion of qualitative methods (oral histories, participant observation, interviews) and the consideration of more user groups (tourism guides and business owners) are needed to establish pathways from participation to value formation – for instance, diving participants may be predisposed to have biocentric worldviews, to the extent that any "increase" may not be expected or measurable by a questionnaire, particularly over a short period of time (1 day) (Smith et al., 2009). Curtin (2005) and Patterson et al. (1998) offer two practical avenues of establishing values through (shark) tourism: (1) the shared experience (e.g., reflecting with other members of a group partaking in the shark dive) can form lasting memories, and perhaps increase the likelihood of value formation through social connection and recalling of personal narratives, and (2) wildlife tourism can offer the traveler to experience "existential authenticity" by giving them an area they are free to explore and project their own personally constructed notions of nature onto. In the latter, the tourist can create their own "sense of place" in the area they explore (potentially encompassing sharks), which may be quite different than their "home" environment where urbanization has led to an increased distance between humans and what is perceived as "authentic" nature. Several studies, however, have suggested a "ceiling effect" with shark tourism, in that the attendees already display pro-environmental attitudes and behaviors, knowledge of sharks, and a strong conservation ethic (Apps et al., 2015; Sutcliffe and Barnes, 2018). While these instances do not denigrate the value of diving with sharks in inciting emotional responses and facilitating the formation of RVs around sharks, they do emphasize the importance of not judging the value of a shark-diving experience by its impact on behaviors, knowledge, or attitudes.

In recent decades, there has been a shift in sharks' public image as "man-eaters" to conservation icons, which has coincided with growing interest to observe sharks in their natural environment. Discourse analysis of a popular Australian diving magazine showed that from 1953 to 2006, there was a shift of "danger-seeking hunter" to "nature-seeking observer" ascribed to shark interactions (Whatmough et al., 2011), although an analysis of newspaper articles concerning sharks showed that both American and Australian outlets reported on the risks sharks pose to humans, more so than the risks human activities pose to many shark populations (Muter et al., 2013). Leveraging sharks for tourism has risen in popularity, although value assessments of this industry have focused largely on financial benefits, for areas including Fiji (Vianna et al., 2011), the Bahamas (Haas et al., 2017), Palau (Vianna et al., 2012), and Australia (Huveneers et al., 2017). Indeed, shark tourism has been proposed as an alternative livelihood in areas where fishing pressure overlaps with shark populations, based on economic valuations which demonstrate how tourism income could supplement loss of fishing income, and assume that a transition from fishing to "non-use" tourism will aid conservation of shark populations (Bentz et al., 2014; Garla et al., 2015; Pires et al., 2016). However, such a proposal must be made on a strictly case-by-case scenario, including evaluation of (1) feasibility of establishing a sustainable tourism business, and (2) whether the amount of fishing pressure in question would have a significant impact on shark populations in question (e.g., species may be highly migratory, and local fishing pressure may be low), and (3) whether increased visitation from growth in tourism could lead to other environmental issues, such as waste management and pollution (Partelow and Nelson, 2018). Records of conflict between shark tourism and fishing businesses (Anderson and Ahmed, 1993; Bentz et al., 2014), and with members of the public concerned about beach safety (Neff, 2014a), point to a need for qualitative studies which capture RV and policy preferences of each stakeholder group, in order to acknowledge and address as many interest groups as possible in the policy creation and implementation process – particularly in areas such as Palau where entire small-island economies are becoming reliant on tourism income (Vianna et al., 2012).

Value Formation Through Media: Emotional Responses and Conservation Ethic

Although shark attacks are extremely infrequent given how often humans use beaches (West, 2011), popular media has instilled and perpetuated a fear of sharks through such fictional movies as Jaws in 1975 (Simpfendorfer et al., 2011; Neff and Hueter, 2013), and the subsequent Jaws 2 (1978), Jaws 3 (1983), and Jaws The Revenge (1987). Although interaction with sharks, whether physical or virtual, presents the opportunity to form RVs, fear-centric media and dialog is likely not conducive to forming RVs around sharks. In this framing, the shark is cast as a villain that challenges human well-being (Muter et al., 2013; McCagh et al., 2015; Sabatier and Huveneers, 2018), rather than part of a positive experience that builds emotional bonds with nature. As recently as 2014, action by politicians in Australia were linked to the fear-based Jaws narrative to mobilize a shark culling (targeted killing) policy despite a lack of empirical

evidence for efficacy (Neff, 2014b; McCagh et al., 2015). Neff and Hueter (2013) have explored origins of the "man eater" image of sharks, an image which may influence RV humans form with respect to sharks. Recently, Pepin-Neff and Wynter (2018) surveyed attitudes toward sharks in Australia, issuing surveys close to the time of shark attacks. They found that an individual's relative feelings of pride, fear, and perception of an attack were intentionality associated with whether the individual was supportive of shark control policies; namely, levels of pride mediated whether or not fear affected policy preference. For instance, at high levels of pride, fear had little effect on policy preference, while the combination of low pride, high fear, and perception of intentionality, was associated with preference for lethal over non-lethal control policies. Simmons and Mehmet (2018) demonstrate the complex effects of implementing policies and monitoring strategies in Australia which address public safety. A survey of social media responses to various non-lethal shark control measures, including those communicating the locations of sharks to the public (e.g., helicopters, shark location-sharing technology) showed that individuals expressed relief or comfort from the knowledge of sharks being monitored, yet also, fear from increased awareness of shark movements. Similarly, Gibbs and Warren (2015) surveyed perceptions of shark control measures in the same region, finding that members of the public were wary of the financial costs of monitoring and controlling sharks for public safety, and were exercising personal agency in taking steps to reduce probability of shark encounters.

In a conservation ethics context, individuals who identify or empathize with the natural environment (or with an animal in that environment), a view which might have been fostered through emotional or shared social experiences in nature may more likely treat that environment or animal as they would treat themselves and other humans – in other terms, extending moral values toward non-humans (Milton, 2002; Nightingale, 2012; Clayton and Susan Opotow, 2013), and displaying behaviors indicating stewardship. For instance, one individual may experience sharks only through media exposure (fictional, news, or documentaries) that depicts sharks attacking humans, describes sharks using negative or criminalizing language, or melodramatically villainizes sharks as menacing through the employment of ominous soundtracks or leitmotifs. (McCagh et al., 2015; Nosal et al., 2016b; Fraser-Baxter and Medvecky, 2018; Sabatier and Huveneers, 2018). In alternative scenarios, an individual may experience sharks through scuba diving (Whatmough et al., 2011; Apps et al., 2015, 2018) and recreational fishing (Lynch et al., 2010; Mcclellan Press et al., 2015; Heard et al., 2016; Shiffman et al., 2017) which engender a familiarity with sharks in the wild; or through documentaries and aquarium exhibits which frame sharks in a positive, non-threatening way (Gendron, 2004; Nosal et al., 2016b; Pepin-Neff and Wynter, 2018). Experiences via contact in the wild or in educational settings where sharks are positively framed are more likely to lead to emotional attachments which engender RV such as stewardship, whereby the individual may be more predisposed to support policies or projects for sustainable shark populations. Although the Emotional Affinity interpretation of value-formation through nature does include such emotions as indignation and anger, these emotions foster a connection in a context of anger for lack of conservation action, for instance, or indignation at environmental degradation (Kals et al., 1999). For sharks, this could be through films showing these animals being killed by fishers, inciting emotions of anger that build a feeling of affinity with sharks and fostering RVs such as stewardship.

Indeed, sharks have risen as an icon of conservation; many species are large and charismatic, prolific in global popular culture, hold human fascination, and are often-touted as symbols and stewards of healthy ocean ecosystems (Simpfendorfer et al., 2011). This popularity has led to campaigns around shark conservation, and are supported by an array of non-governmental organizations, individuals, and other actors. Although shark conservationists have not been specifically studied as a group, looking for scholarship in the environmentalism movement can shed light on the formation or strengthening of RVs through engaging in the shark conservation movement. Some individuals might identify as an environmentalist through simple day-to-day actions such as recycling, and feel a sense of social/collective identity with other environmentalists as a result of their actions (Markle, 2014; Mccright and Dunlap, 2015; Nelms et al., 2017). Companies may institute "eco-friendly" policies or practices to capitalize on positive social associations with environmentalism (Wry and York, 2017). Non-governmental organizations, petitions to decision makers and companies, and now social media, are avenues for individuals to engage with scientists and decision makers (Yang, 2005; Leeder, 2007). With respect to sharks, a "shark conservationist" environmentalist social identity is likely to be facilitated by similar forms of engagement to exercise agency. For instance, individuals who identify as part of a conservation group can score higher on measures of emotional connection with nature, potentially showing a "positive feedback" effect through engagement in conservation action (Kals et al., 1999). Social media in particular is emerging as a powerful tool for education professionals and scientists to communicate with the public, to translate the engagement of "environmentalist" identifying individuals into conservation outcomes (Parsons et al., 2014b).

Education: Has Knowledge Provisioning Led to Stewardship?

For many, aquariums can offer a more accessible location than the wild to form RV through the observation of sharks, even though they are in a human-created rather than natural environment. Over time, aquariums have increasingly framed sharks as objects of conservation concern and ecological importance, although a few retain the element of "sensationalizing" them to draw interest from visitors (Gendron, 2004). For example, "Shark Dive Xtreme" at Melbourne Aquarium in Australia, which offers visitors the chance to swim with sharks, could be interpreted as sensationalizing the notion of human-shark interaction; others, though, present the opportunity to swim with their sharks in a more neutral manner, like "Sea Swim" at the Florida Aquarium in the United States. Indeed, it is difficult to make generalizations

regarding aquarium exhibits of sharks and their impacts; a self-reporting questionnaire in the United Kingdom found aquarium attendance to be associated with positive attitudes toward sharks (Friedrich et al., 2014), while another study found that although childhood visitors identified the utilitarian value of sharks in their environment, pre-existing notions about sharks attacking humans, and sharks being an extractable "resource" to be harvested (e.g., an instrumental valuation) persisted throughout the visit (Correia das Neves and Rocha Monteiro, 2014).

Knowledge, or information provisioning alone, is often insufficient to build an individual's conservation ethic, or their drive to preserve natural resources for future generations; the information presented may be incongruent with personal ideologies or mental models of how the world works, as with climate change denial (Sterman, 2008; Jacquet et al., 2014), or an experiential complement may be necessary, in which the individual directly interacts with the natural environment or animal they are learning about (Otto and Pensini, 2017). Education practitioners have recognized the importance of "experiential education" – educational programming in nature (Stern et al., 2008) or "hands-on" learning opportunities in the field (Manzanal et al., 1999) which has the potential to enhance comprehension of subject matter, and increase pro-environmental attitudes among students. The relative contributions of experience and knowledge are intractable, whether in an ecotourism (experience-focused) (Ballantyne and Packer, 2002; Powell and Ham, 2008) or educational (knowledge-focused) setting, however, strengthening their combination appears more likely to engender RV than one component alone (Pepin-Neff and Wynter, 2018), particularly when there are "follow-up" activities after the initial experience (Hughes et al., 2011; Behrendt and Franklin, 2014).

Shark Researchers: Science as a Practice to Form Relational Values

Scientists hold a pivotal role in generating knowledge for the valuation of sharks, whether economic (evaluating financial costs and benefits from different uses of sharks for livelihood support), intrinsic (demonstrating ecological importance which encompasses sharks in an eco-centric ideology), or relational (demonstrating the complexity of sharks' behavior to contrast "man-eater" perceptions, and generation of knowledge in collaboration with fishers). Apart from a scientist's well-being relating to sharks by employment to conduct research, the act of studying sharks and disseminating this knowledge may contribute to personal and community identities (the scientist and the scientific community). As an interesting duality, scientific knowledge has itself been discussed as a type of socio-cultural ES (Costanza et al., 1997). While no studies have aimed to investigate RVs of scientists, an historical overview of shark researchers and institutions reveals that their research has evolved in parallel with how sharks are used (and viewed) by society – from expendable resources of the sea (Salviani, 1554; Castro, 2014) to a threat to human ocean users (Klimley, 1974; Zahuranec, 1975; Gilbert, 1977) to a potentially critical species for maintaining ecosystem functionality (Heithaus, 2001; Navia et al., 2014) to icons for marine conservation (Simpfendorfer et al., 2011; Whatmough et al., 2011) to critical elements of socio-economic systems. If scientists have developed an "emotional affinity" for their study subjects, and are more likely to display pro-environmental behavior in the context of elasmobranch conservation measures, it is important for them to examine whether this presents a personal bias toward certain management measures, and make an effort to examine all possible management options rather than what is perceived as the "best" management option from their own lens (Kiik, 2019). The progression of elasmobranch research has occurred within a broader social context, including changes in the philosophy of science and the structure of academic institutions, revolutions in scientific understanding, and larger societal events that influenced funding of research (Kuhn, 1970; Kindi, 1995). Each particular researcher undertakes their investigations from the lens of a certain worldview, being influenced by a unique combination of mentors, colleagues, ideas, and environments. The present status of elasmobranch science, as with any field of research (social science being the most self-reflexive in the published literature, e.g., Hammersley and Gomm, 1997; Hammersley, 2005, 2006; Wilholt, 2009), is a result of this dynamic and complex set of interactions. Given the wealth of research tools and methodologies, and emergence of interdisciplinary teams of researchers to tackle complex conservation issues, elasmobranch scientists are well-positioned to evaluate the most effective investment of human and financial capital, to pursue socially and ecologically impactful research questions versus being bound to taxonomic or methodologically biases (Huveneers et al., 2015).

Roots of shark science with the beginnings of natural history

The acceleration of shark research over the past century is preceded by millennia of human inquiry of the natural world. Through the value of their flesh, liver oil, and skin, elasmobranchs have been a resource of food and raw materials for humans for thousands of years (e.g., instrumental and economic value). In European writing of sharks from the 1500s, these animals were portrayed as irritable, purportedly harassing of fishers who would dive for their catch of non-elasmobranch species (Salviani, 1554; Castro, 2014). The first records of elasmobranchs in the realm of natural science appeared around 384 B.C.E. in the writings of Aristotle (Castro, 2014), namely aspects of anatomy and development (including the first exposition on animal embryology), and behavior. Continuing through the 20th century, elasmobranchs were studied and described from these instrumental and biological positions: several men educated in the medical arts would contribute to the written knowledge of elasmobranchs, including illustrations and texts thereof as part of larger works. Ippolito Salviani, in his "Aquatilumanima" (Salviani, 1554) describes not only the species' appearance, but also culinary preparations – suggesting that these fish were utilized for subsistence and/or as an economic resource, and that the natural scientists at the time considered this knowledge important to convey. The concept of a "natural history" is thought to have begun with Pliny the Elder (A.D. 21/24 – 79), who opined that nature was present to serve man. When composing the natural history of a region, he included not only animals and plants, but also geological and cosmological aspects

(Jashemski and Meyer, 2002). It may follow that natural scientists (and natural historians) were compelled to describe how animals were useful to humans given that was their reason for existence, demonstrating their instrumental value alongside biological or ecological observations.

Shark research co-evolves with the institution of science

As scientific inquiry as a whole evolved to become systematic, so did observations of elasmobranchs. Technological advances and societal expansion led to more elasmobranchs being discovered, and the 1900s saw a drastic increase in shark studies in the United States, while they were both used as a human commodity, and the capacity grew to specifically seek out sharks in their natural environment. With World War II, the US Navy began funding research to prevent shark attacks, providing considerable resources and an impetus for scientists to systematically study sharks and develop the basis for present day institutions and research (Castro, 2017). This research continued after the war, with the US Navy's interest in understanding how to protect their personnel from sharks providing financial backing for much foundational research in shark behavior and sensory biology (Tester and Kato, 1963; Gruber et al., 1975). In fact, these efforts led to the inception of the first collaborative shark research group, the AIBS Shark Research panel. Scientists had identified that ". . .if better methods for protection against shark attack are to be developed, a broad program of basic scientific research will have to be instituted," also noting that out of the 350 species at the time, "only two dozen are considered to be dangerous, and a still smaller number are listed to be regularly dangerous" (Aronson and Gilbert, 1958). During the 1958 panel meeting, members stated the need to improve species identification (basic morphology and traits), knowledge of ecological and geographic preferences, and scientifically rigorous observations of behavior. With respect to the latter, there was an urge to adopt systematic methods of animal behaviorists rather than putting forth suggestive statements in papers without sufficient evidence. The AIBS Shark Research Panel convened until 1970, producing more than 100 studies. While the guiding premise of this group was protecting humans from sharks, there was an undeniable side effect of vastly expanding the scientific body of knowledge on elasmobranchs. Since the 1970s, shark research has seen vast growth through an influx of governmental and non-governmental financial support.

Research on shark population management, whether for sustainable extraction or recovery, is commonly performed by government agencies (such as the U.S. National Marine Fisheries Service (NMFS) arm of the National Oceanic and Atmospheric Administration (NOAA) which oversees fisheries management planning), affiliated academic institutions (such as the Virginia Institute for Marine Science which conducts shark population surveys in coordination with NOAA), and projects carried out at academic institutions or research centers with governmental and non-governmental financial support. Other avenues of shark research have been important in shaping opinions of sharks as more sophisticated than mere "man-eaters" (Simpfendorfer et al., 2011); for instance, the discovery of sharks' ability to learn tasks (Clark, 1959), navigate long distances through smell and magnetic-field perception (Klimley, 1993; Nosal et al., 2016a), and their support in small-island economies (Anderson and Ahmed, 1993; Vianna et al., 2012).

Harnessing RVs in Research and Conservation

Fostering Stewardship Through Education

If the goal of an aquarium or educational program is to foster positive attitudes toward sharks, more targeted exhibits and curricula focusing on conceptual change and knowledge restructuring should be considered, with ongoing evaluation of how these strategies affect visitors' or students attitudes (Thompson et al., 2002). Although it is important to demonstrate the ecological, instrumental, and intrinsic values of sharks, aquariums can engineer opportunities to form RV, such as through the Value-Belief-Norm or Emotional Affinity frameworks and based on the environmental psychology literature. For instance, through shark-centric activities and events which involve socializing in groups, reflecting on shared experiences or impressions of sharks through writing, emotional engagement through art (Edwards et al., 2016) and storytelling (Woodhouse, 2011), or emphasis on the "natural beauty" of sharks and the ecosystems they support (Zhang et al., 2014).

In any educational setting, it is important to impart the role of sharks to society as well as to natural ecosystems, so that conservation attitudes are well informed and not counter-productive. For instance, students taught about the whale hunting practices of the Makah Tribe in Washington State (United States) did not learn of the cultural importance of whale hunting, and were imparted with negative attitudes toward the tribe (Marker, 2006). Rather, educators can use these opportunities to foster sensitivity and awareness of the diversity of worldviews, cultures, rather than a "protect at all costs from other users" approach to conservation.

Engagement with sharks by virtue of their iconic appeal may present a "gateway" to engagement with environmental issues, apart from overfishing, which are affecting sharks. For instance, climate change, habitat degradation, and marine pollution are pressing and challenging issues in global marine conservation (Parsons et al., 2014a), which could be illustrated in an educational setting through their direct effects on sharks. However, we must caution against a one-dimensional "flagship species" approach that marginalizes other, less charismatic species, which are at equal or greater risk of extinction (Dulvy, 2013; Liordos et al., 2017; Curtin and Papworth, 2018). Rather, such a "gateway" approach to educational programming might begin with sharks, and end with broader impacts to other species, ecosystems, and human communities.

Quality of Life Through Restoring "Lost Connections" to Values

The importance of recognizing linkages among culture, and mental and physical well-being are underscored by contemporary challenges in improving health measures in Aboriginal groups worldwide [e.g., Canada (Bennett et al., 2018) and Australia (Burgess et al., 2005), or see Stephens et al., 2006; Axelsson et al., 2016 for global reviews]. Ahead of colonialization,

the dependency of these groups upon a healthy environment was reflected in a strong stewardship ethic, via active management of their natural resources. When this role was adopted by a colonial government, and now the present-day administration, these indigenous groups lost a critical cultural connection through the curation of their support systems, which were linked to not only sustenance but cultural identities and mental well-being (Burgess et al., 2005). By documenting these losses of access and rights, present-day scientists, managers, conservation practitioners, and policy-makers can identify entry points for improving well-being along with resource management, by way of restoring these "lost connections" to values, as much as is possible in the current social and ecological context.

Reflexivity in the "Natural Sciences"

At present, while shark research activities span the global oceans, the United States, Australia, and the United Kingdom host the majority of academically associated research institutions (**Figure 1**). Elasmobranch research, rather than a discipline in itself, borrows from fields such as fisheries science, physiology, behavior studies, biological oceanography, and many others. As such, paradigms of these other fields, as well as general evolutions in science and its philosophy, permeate elasmobranch research and its practitioners. In the social sciences, the implicit bias of humanity is acknowledged through the concept of reflexivity. However, the need for self-reflection to overcome human biases is unique to no field of study, given that humans administer the research, and a history of any discipline will show transformations of paradigms and modes of thought (Kuhn, 1970). Acknowledging this tendency through the practice of reflexivity can allow researchers to innovate their approaches to institutionally instilled modes of inquiry.

Conflicts Among Resource Users: Using Relational Values as a Pathway to Mediation

A common theme that emerged from commercial fishers is that they are perceived as the "enemy" of tourism or he recreational fishing of sharks (Anderson and Ahmed, 1993; Shiffman et al., 2017). Numerous publications are available to appropriately critique cumulative impacts of shark fisheries (and shark bycatch); these studies show that while overexploitation is associated with population declines, appropriate management can lead to rebounding or sustained shark populations, and sustainable shark fisheries (Campana et al., 2006; Dulvy et al., 2014; Peterson et al., 2017; Simpfendorfer and Dulvy, 2017). The idea that shark fishing and shark products can provide all or (more commonly) a portion of an individual's income (Jabado et al., 2015; Jaiteh et al., 2017), was supported by a survey of shark researchers, as long as the fishing was sustainably managed (Shiffman and Hammerschlag, 2016). Further to these points, shark fishing likely contributes to the fishers' identity and other non-financial aspects of well-being; accordingly, based on this review, we suggest that effective negotiation for sustainable management of shark populations necessitates an open-minded stance toward the perceptions and values of individual commercial fishers, and cautions against generalizing all commercial efforts as detrimental, or assigning blame at the individual level. This type of approach is similar to "moral relativism," in which moral judgements are formed by a group of people (i.e., fishers, managers, scientists, etc.), and those collectively held moral positions "make sense only in relation to and with reference to one or another" (Harman, 1975).

In fact, assigning blame or framing shark conservation as a fishery-led problem may be counterproductive in effecting policy for shark population recovery or sustainability, in part due to this framing not considering the full suite of values derived from sharks. Healthy fish/shark populations are supported by both "environmentalists" and fishers, however, "Fishers find it difficult to relate to environmental movements because they are the ones cast as the problem" (Nightingale, 2012). In reality, the current state of a given shark population is the result of large-scale, complex and historically precipitated geopolitical and economical dynamics (Finkbeiner et al., 2017). A more constructive dialog acknowledges the importance of sharks to all parties involved, and if reduced fishing pressure is needed, seeks to conserve relationships and their derived values as much as possible. This collaborative approach can yield unexpected knowledge, solutions, and compromises (Klain and Chan, 2012; Daw et al., 2015; Oteros-Rozas et al., 2015).

Groups with conflicting views tied to values or morals (e.g., managing for tourism versus for commercial fishing, unqualified support for banning shark fishing from environmental groups versus supporting sustainable shark fishing as a source of income), can lead to a polarizing dialog, and barriers to negotiation and enforcement of policy for managing the resource (sharks) (Biggs et al., 2017). Convening stakeholders to find areas of common ground, such as around shared values of sharks, as part of a collaborative, iterative, and transparently evaluated process, can build trust and provide a pathway for mediating conflicts (Kahane, 2012) and charting a course for sustainability which accounts for the multi-faceted value of sharks.

How Can Assessments Incorporate Relational Values From Sharks?

Value Definition for Clarity in Decision Making and Knowledge Exchange

To facilitate the incorporation and recognition of RVs, it is important to have a common language of what constitutes value types in general. While there have been no lack of studies on the economic (financial) values of sharks, these economic values are sometimes referred to as "social" benefits. However, monetary values, particularly if focused on a specific industry (e.g., scuba diving), are not a substitute for other values a society derives from sharks. We suggest a simple terminology following Small et al. (2017), where a value or ES falls under one (or several) of the following framings:

(1) Ecological: Non-anthropocentric, functional role in the ecosystem (e.g., nutrient cycling, algal grazing)
(2) Economic: Monetary or financial (e.g., contingent valuation, economic benefit, revenue)
(3) Socio-cultural: Non-monetary benefits, practices, and goods, which may be categorized as:

- Instrumental: Of direct use to humans (e.g., a coral reef for recreational use)
- Intrinsic: Of value in and of itself (e.g., pristine coral reefs)
- Relational: Resulting from a relationship with nature

A Workflow for Conducting Holistic Values Assessments

Rather than advocating for one type of valuation over another, we aim to show that non-monetary values of elasmobranchs and their associated coastal ecosystems can complement monetary valuations of sharks which have emerged in the literature. The most effective combination of value types in a given assessment/valuation is context-dependent, and each value type has its own merit. In this vein, we propose a flexible, inclusive, and outcomes-focused workflow for eliciting and incorporating values around shark conservation issues, from the scale of local communities to international negotiations (**Tables 1**, **2**), similar to Pascual et al. (2017). Assuming a commonly held goal of sustainably managed shark populations, economic values by themselves are often insufficient, or even misleading and detrimental, in the quest to achieve this goal. Valuations or assessments of an elasmobranch resource - whether a local population, fishery, or associated dive industry – should instead take a bottom up approach that allows for all subjective values, including economic values if appropriate for the scenario.

Methods to Elicit RVs

Taken alone, surveys or questionnaires may be insufficient to elicit RVs, however these methods can be part of an

TABLE 1 | Across increasing demographic scales, examples of methods to elicit holistic values of individuals and groups, and specific considerations for assessments at that demographic scale.

Scale	Example assessments or methods	Specific considerations
Individual – community	Ethnography, "deliberative methods" (discussion groups, workshops)	Create "opportunities for individuals to express, exchange, reflect, negotiate, and develop their views and evidence in response to those of others"; consider intergenerational dimensions of values
Community – region	Key stakeholder interviews paired with surveys/questionnaires	Heterogeneity of norms among communities, yet critical for revealing conflicts; time/resource intensive to compose holistic indicators for scaling
Regional – national	United Kingdom National Ecosystem Assessment	Incorporate indicators from finer-scale assessments, or perform case studies for scaling (allow for time/resource intensive process)
International	Millennium Ecosystem Assessment (MEA), Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES)	Heterogeneity of norms toward and definitions of ES, prioritization of ES, and buy-in among assessment leaders, users, and parties being assessed; caution with generalizing results between socio-cultural contexts and demographic groups

TABLE 2 | Guiding workflow to conduct a holistic values assessment, whether as an independent study, or as a component of a longer-term planning/implementation process.

Assessment stage	Guiding questions
1. Goal-setting	• What are the researchers' goals of the assessment or valuation?
2. Preliminary scoping of stakeholders	• Are there local conservation or government/management initiatives operating on the same resource or issue? How do their goals or actions complement/contrast with those of the researchers? • Who will be receiving results of the assessment? Is the assessment part of a long-term conservation/management planning and implementation process, or could it provide added value to an existing similar process? • Resource users: how do people use the resource themselves? how do they think others use it? • Perceived value types: do they think it is important for themselves or the community – why or why not? has it been valuable in the past? how do other people talk about the resource?
3. Values elicitation	• Semi-structured interviews, oral histories, focus groups, or participatory mapping activities to elicit and record all perceived values of all relevant stakeholders. Address: role of the resource in identity and sense of place, social activities, emotional attachments, well-being • If appropriate, economic valuation of the resource (potentially restricting to certain user groups to avoid detrimental effects of assigning monetary values)
4. Synthesis, application, and communication	• How do stakeholders relate to the resource? Do different groups have different value framings (i.e., economic versus relational)? • Have there been any changes in "value" over time? Are there disparities among user groups (tourism operator, fisher, other local resident) or by demographic categories (gender, income level)? • If part of a longer-term planning/implementation process, consider how "indicators" of value or well-being could be repeated in future monitoring studies. • Based on goal-setting, deliver results of the assessment to relevant institutions and individuals – results can be framed with respect to the goals and ideologies of the recipient by emphasizing the most relevant value types.
5. Long-term monitoring (if applicable)	• If part of a longer-term planning/implementation process: repeat indicator/value/service assessments. • Consider: how are stakeholder groups evolving over time? (e.g., relative size, influence, access to resources, access to livelihood diversification)

TABLE 3 | Cases from the literature which assessed or described the "value" of elasmobranchs to a community, and/or elicited relational values toward a coastal marine resource.

Values (given)	Value type(s)	Method(s) to elicit values	Stakeholder group(s)	Region	Species/ Ecosystem	Reference
Non-elasmobranch focus						
Well-being	SC-R	Questionnaire for self-assessment of "well-being" (definition and drivers)	Commercial fishers	Queensland, Australia	Fisheries of Cooktown, Port Douglas, Innisfail, Townsville, and Bowen	Marshall and Marshall, 2007
Non-monetary/ intangible/ cultural, and monetary value	SC-IC, SC-IL, SC-R, E	Semi-structured interview (identify values), mapping values (monetary and non-monetary) and threats (indicate location)	Fishers, managers ("marine reliant professions")	Northern Vancouver Island, Canada	N. Vancouver Island seascape	Klain and Chan, 2012
Ecosystem services: fishery materials, habitat, coastal protection, sanitation, recreation, bequest, education, cultural	SC-IC, SC-IL, SC-R	Questionnaire (ranking importance of coral reef ecosystem services, suggestions for improvement of services)	Fishers, fish traders, fish factory workers	Madagascar, Tanzania, Kenya, Seychelles	Coral reefs of W. Indian Ocean	Lau et al., 2018
Conservation values and behavior, willingness to pay for conservation	E	Questionnaire (impact of educational content on values/ behavior, contingent valuation)	Park visitors	Queensland, Australia	Turtles of Mon Repos Conservation Park	Tisdell and Wilson, 2002
Indigenous cultural values, moral principles around nature	SC-IC, SC-IL, SC-R	Media review, literature review	Makah tribe members, media, Port Townsend students, teachers, parents	Washington, United States	Whales in traditional fishery	Marker, 2006
Intrinsic, instrumental, new ecological paradigm, and relational values	SC-IC, SC-IL, SC-R	Questionnaire (response to different value frames: intrinsic, instrumental, new ecological paradigm, and relational values)	NE United States public, Costa Rican farmers, tourists to Costa Rica	NE United States, Costa Rica		Klain et al., 2017
Environmental attitudes and non-use values (willingness to pay for conservation)	E, SC-IC, SC-IL	Questionnaire (contingent valuation, new ecological paradigm)	Random sample of state residents	Maine, United States	Peregrine falcons, shortnose sturgeons	Kotchen and Reiling, 2000
Willingness to pay for sustainable seafood	E	Questionnaire (contingent valuation, environmental attitudes and beliefs, trust of certification, use of ecosystem)	Fishers (reef anglers)	Florida, United States	Florida Reef tract	Harper, 2015
Well-being, work/ income, job satisfaction, social networks (SC-R)	SC-R, SC-IL, E	Oral history (interview)	Commercial/ professional fishers	Maine, Massachusetts, Rhode Island, New York, and New Jersey, United States		Colburn and Clay, 2012
Cultural values, life themes: relationships with people, and with nature, spiritual and religious beliefs	SC-R, SC-IL, SC-IC	Oral history (interview)	Female elders (kupuna)	Oahu, Hawaii, United States	Coastal Hawaii, United States	Mokuau and Browne, 1994

(Continued)

TABLE 3 | Continued

Values (given)	Value type(s)	Method(s) to elicit values	Stakeholder group(s)	Region	Species/ Ecosystem	Reference
Social importance: interactions, hunting, use of parts, stories/legends, beliefs, conservation attitudes	SC-IC, SC-IL, SC-R	Questionnaire via interview (closed and open-ended questions) administered in group setting	Villagers (residents and leaders)	Andaman Coast, Thailand	Dugongs and associated mangrove/seagrass ecosystems	Hines et al., 2005
Job satisfaction, well-being, self-actualization	SC-R	Semi-structured interview, questionnaire	Commercial fishers	Northeast United States	Fisheries o NE United States	Pollnac et al., 2015
Community, kinship, crew connections, adventure and money	SC-IL, SC-R	Oral history, social network analysis	Commercial fishers	Newport, Alaska, United States	Fisheries of Newport and Kodiak, Alaska	Package-Ward and Himes-Cornell, 2014
Place attachment (identity and dependence)	SC-R	Self-administered questionnaires	Coastal community residents (<10 and >10 miles from various marine reserves)	Oregon, United States	Marine coast of Oregon, United States	Perry et al., 2014
Cultural ecosystem services, well-being, "the good life"	SC-IC, SC-R	Participant observation, interviews, participatory workshop	Fishing community residents	Lofoten Islands, Norway		Kaltenborn et al., 2017
Fishing dependency/importance of fishing: economically, socially (identity, sense of community)	SC-IL, SC-R	Open ended interviews, "contextual" and historical background information	Fishers (commercial and recreational), individuals with businesses related to fishing, community officials and leaders	Galveston Bay Complex, United States	Fisheries near Galvelston Bay, San Antonio Bay	Jacob et al., 2010
Elasmobranch focus						
Willingness to pay, willingness to donate for conservation	E	Questionnaire (contingent valuation, attitudes and beliefs toward wildlife, prior donation behavior)	Tourists (domestic and international)	Galapagos Islands, Ecuador	Scalloped hammerhead sharks, sea turtles	Cárdenas and Lew, 2016
Biocentrism	SC-IC, SC-IL	Questionnaire (knowledge and biocentrism), two treatments (with/without educational talk)	Tourists	New South Wales, Australia	Gray nurse sharks	Smith et al., 2014
Cultural, ecological	SC-IC, SC-IL	Archaeology (archived shark weapons, museum collection for regional marine species), literature review (historical texts)	Gilbertese Islanders	Republic of Kiribati, Central Pacific	Shark communities near Gilbert Islands	Drew et al., 2013
Customs and beliefs toward sharks	SC-IC, SC-IL, R	Interviews, historical texts	Gilbertese Islanders	Republic of Kiribati, Central Pacific	Gilbert Islands	Luomala, 1980
Perceived ecological value of sharks and protected areas, concern/emotions toward declining shark populations	SC-IL, SC-IC, SC-R	Interviews (guided by questionnaire)	Fishers, non-fishing community members (cultural leaders, teachers, shop owners, traditional healers, leaders of community organizations)	Raja Ampat, Indonesia	Shark fisheries of West Papua, Papua and Maluku provinces, Indonesia	Jaiteh et al., 2016b

(Continued)

TABLE 3 | Continued

Values (given)	Value type(s)	Method(s) to elicit values	Stakeholder group(s)	Region	Species/ Ecosystem	Reference
Livelihood perceptions: financial viability	SC-IL, E	Interviews (guided by questions), participant observation	Shark fishers (active and retired), non-fishing community members, fishing bosses	Eastern Indonesia	Shark fisheries of Osi, Dobo and Pepela, Indonesia	Jaiteh et al., 2017
Value of sharks to dive tourism industry: economic benefits, welfare (income)	E	Questionnaire (contingent valuation)	Dive operators and tourists	Republic of the Maldives	Dive sites of the Republic of the Maldives	Zimmerhackel et al. (2018)
Value of shark diving industry to finance marine protected area	E	Questionnaire (contingent valuation)	Dive tourists	Semporna, Malaysia	Dive sites near Semporna	Vianna et al., 2017
Value of shark diving: local income, GDP, taxes	E	Scoping interviews, questionnaires (financial expenditure)	Divers, dive operators, dive guides, local fishers	Republic of Palau	Dive sites near Republic of Palau	Vianna et al., 2012
Economic value of shark diving tourism	E	Questionnaires (financial expenditure)	Dive tourists	Australia	Whale sharks, white sharks, gray nurse sharks, reef sharks	Huveneers et al., 2017
Perceived ecological and economic importance (value)	SC-IL	Questionnaires, website content analysis	Shark fishing charter boat operators	Florida, United States	Coastal FL	Shiffman and Hammerschlag, 2014
Economic value of sharks and rays through tourism, media, research, conservation	E	Structured interviews with dive operators, questionnaires (financial expenditure), online surveys (filming or research expenditures)	Dive operators, dive tourists, media groups, researchers/research organizations	Bahamas	Bahamas coastal and surrounding marine areas	Haas et al., 2017
Social and economic benefits from sharks (via tourism, fishing, taxes, and fees)	E	Questionnaires (financial expenditure, interest in seeing sharks)	Tourists, restaurant owners, dive operators, photographers, dive guides, fishers, environmental management authorities	Fernando de Noronha, Brazil	Archipelago of Fernando de Noronha	Pires et al., 2016
Value of shark diving for emotional engagement, conservation behavior, knowledge	SC-R, SC-IL, SC-IC	Questionnaires	Dive tourists	Neptune Islands, South Australia	White sharks	Apps et al., 2018
Economic benefits from conservation measures	E	Questionnaires (contingent valuation)	Tourists at marine resort	Yasawa Islands, Fiji	Reef manta rays	Murphy et al., 2018
Social and economic importance of shark fishery to community	E, SC-IL	Semi-structured interviews, questionnaire, participant observation	Shark fishers, shark fin collectors	Toliara, Madagascar	Fisheries of Soalara, Beheloka, Maromena, Befasy	McVean et al., 2006
Beliefs and attitudes around shark tourism	SC-R, SC-IC, SC-IL	Questionnaires with open ended questions (theory of planned behavior)	Shark diving tourists	Neptune Islands, Australia	White sharks	Apps et al., 2016
Educational and conservation benefits of shark tourism	SC-IL, SC-IC	Questionnaires (perceived ecological and economic value, knowledge of sharks and conservation issues)	Tourists and residents of Fernando de Noronha	Fernando de Noronha, Brazil	Archipelago of Fernando de Noronha	Garla et al., 2015

(Continued)

TABLE 3 | Continued

Values (given)	Value type(s)	Method(s) to elicit values	Stakeholder group(s)	Region	Species/ Ecosystem	Reference
Value of shark diving to tourism industry	E	Questionnaires (financial revenue, motivation for visit, shark knowledge, dive experience)	Tourists of Azores	Azores archipelago	Dive sites near the Azores	Torres et al., 2017
Shifting value of shark fishing (annual price per kilogram)	E	Fishery data from central auction house and Azores commercial fishery authority	Commercial shark fishery	Azores archipelago	Azores Exclusive Economic Zone	Torres et al., 2016
Monetary value of sharks through tourism	E	Questionnaire (influence of sharks in decision to dive, financial expenditures)	SCUBA dive operators, tourists (divers and non-divers)	Moorea, French Polynesia	Sicklefin lemon sharks of "Opunohu" dive site	Clua et al., 2011
Values and attitudes underlying attitudes to policy	SC-IC, SC-R	Social media (sentiment analysis), Focus groups (coded by "affect," or emotion, toward capacity, propriety, value, composition, normality)	Surfers, swimmers, lifesavers, small business owners, tourism operators, conservationists, anglers	New South Wales, Australia	New South Wales coastal marine ecosystem	Simmons and Mehmet, 2018
Knowledge and attitudes toward sharks: pride, blame, fear	SC-IC, SC-R	Questionnaire following 1 of 3 priming "treatments" (pre and post aquarium visit)	Aquarium attendees (shark exhibit)	☐SEA LIFE Sydney Aquarium, New South Wales, Australia	New South Wales coastal marine ecosystem	Pepin-Neff and Wynter, 2018

Values are listed as described in the study, whether the value types fall under the socio-cultural-intrinsic (SC-IC), -instrumental (SC-IL), relational (SC-R), or economic (E), framings as described by Small et al. (2017).

integrated approach; for instance, at the scoping stage of an assessment: (1) to gather socio-economic data such as income, occupation and age; (2) by asking participants about interactions with other members of the community, to identify key stakeholders or "knowledge brokers" for in depth interviews; and (3) if snowball sampling is desired, by including a field that indicates other stakeholders to include in the assessment (Nayak, 2017). Snowball sampling, where participants identify further individuals or groups which are relevant to the study (Atkinson and Flint, 2001), is useful to expand the scope of the assessment, and reveal social networks through peer to peer recommendations. Any sampling methodology, however, has inherent limitations, so researchers may consider integrating multiple sampling strategies to account for varied groups of participants or stakeholders, including targeted sampling where populations might be entirely missed by traditional methods (Watters and Biernacki, 1989), thereby minimizing bias in the study. Importantly, scoping can identify barriers to an inclusive assessment (e.g., institutional, cultural, capacity) at the early stages, allowing the researchers time to refine their methods. Large-N surveys or questionnaires can also be used after values have been elicited from in-depth interviews or focus groups, to (1) gauge whether the same values are recognized on a larger scale, (2) whether framing conservation messaging in the language of the elicited RV is more or less resonant than other framings (Kusmanoff et al., 2016; Klain et al., 2017), or (3) gauge preferences of stakeholders for alternative conservation plans, with these alternatives based on the initial values assessment (Etxano et al., 2015). **Table 3** compiles the methods for (1) eliciting RV of marine resources, and (2) assessing values of sharks, relational and otherwise.

Combining Value Frameworks in an Assessment

While all value frames present different perspectives on the same "issue," and are not necessarily intercomparable, it can be important to asses all points of view to realize and communicate tradeoffs in management decisions or consequences of resource declines, and also in appealing for behavior change and buy-in to the results and application of the assessment (locals, policy makers, other stakeholders). If values are reflective of underlying motivational goals, as outlined by Schwartz and Bardi (2001), it is only by allowing full expression of values by a person or group that underlying motivations of behavior toward a resource – now and in the future – can be understood. Alternatively, it may be responsible to restrict certain forms of valuation from an assessment in order not to alter relationships with nature in the course of that valuation.

As the scale of a valuation/assessment increases, so does the range of the values spectrum; although these differences could be perceived as a conflict, and practitioners may choose to

focus on areas of agreement rather than disagreement, cataloging these differences in assigned value or relative importance is an important role of ES assessments (Schwartz and Bardi, 2001). Including temporal and intercultural scales in an assessment can be very revealing; for instance, identifying which values are most resonant over time and among different cultures can highlight shared principles or framings which are both important to emphasize and acknowledge in international negotiations, and appropriate to incorporate in large-scale assessments.

CONCLUSION

From its introduction in 2015, explicit studies of RV have become more common, however, the field remains limited in scope. To gain traction in values assessments, and recognition of their importance in conflict resolution around natural resources, RV must become more accessible as a concept and a tool. The concept of people deriving values from a relationship with nature, and these values being important for their well-being, is intuitive yet abstract. This abstraction can be shifted to clarity through the sharing of case studies of RV across contexts – whether fishery, coastline, or forest management; urban or rural setting; and for any number of outcomes from spatial design of a protected area, increasing compliance with existing management measures, or simply to monitor stakeholder attitudes. These examples can allow practitioners in academic and non-academic spheres to visualize how RV might be leveraged in their work, in new and perhaps unexpected ways.

AUTHOR CONTRIBUTIONS

RS led the conceptualization, research, and composition. All authors contributed substantially to the framing, writing, and editing of the text.

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frontiers in Marine Science

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From Science to Evidence – How Biodiversity Indicators Can Be Used for Effective Marine Conservation Policy and Management

Abigail McQuatters-Gollop[1], Ian Mitchell[2], Cristina Vina-Herbon[3], Jacob Bedford[1], Prue F. E. Addison[4], Christopher P. Lynam[5], P. N. Geetha[6], Estee Ann Vermeulan[7], Kaylee Smit[7], Daniel T. I. Bayley[8], Elisabeth Morris-Webb[9], Holly J. Niner[10] and Saskia A. Otto[11]*

[1] Centre for Marine and Conservation Policy, University of Plymouth, Plymouth, United Kingdom, [2] Joint Nature Conservation Committee, Aberdeen, United Kingdom, [3] Joint Nature Conservation Committee, Peterborough, United Kingdom, [4] Interdisciplinary Centre for Conservation Science, Department of Zoology, University of Oxford, Oxford, United Kingdom, [5] Centre for Environment, Fisheries and Aquaculture Science, Lowestoft Laboratory, Lowestoft, United Kingdom, [6] PG and Research Department of Zoology, Mahatma Gandhi University, Kottayam, India, [7] Institute for Coastal and Marine Research, Ocean Sciences Campus, Nelson Mandela University, Port Elizabeth, South Africa, [8] Centre for Biodiversity and Environment Research, University College London, London, United Kingdom, [9] School of Ocean Sciences, Bangor University, Menai Bridge, United Kingdom, [10] Faculty of Engineering, University College London, Adelaide, SA, Australia, [11] Institute of Marine Ecosystem and Fishery Science, Center for Earth System Research and Sustainability, University of Hamburg, Hamburg, Germany

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Edited by:
Leslie Cornick, Eastern Washington University, United States

Reviewed by:
Jose M. Fariñas-Franco, National University of Ireland Galway, Ireland
Silvia De Juan Mohan, Institute of Marine Sciences (ICM), Spain

***Correspondence:**
Abigail McQuatters-Gollop
abigail.mcquatters-gollop@plymouth.ac.uk

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Indicators are effective tools for summarizing and communicating key aspects of ecosystem state and have a long record of use in marine pollution and fisheries management. The application of biodiversity indicators to assess the status of species, habitats, and functional diversity in marine conservation and policy, however, is still developing and multiple indicator roles and features are emerging. For example, some operational biodiversity indicators trigger management action when a threshold is reached, while others play an interpretive, or surveillance, role in informing management. Links between biodiversity indicators and the pressures affecting them are frequently unclear as links can be obscured by environmental change, data limitations, food web dynamics, or the cumulative effects of multiple pressures. In practice, the application of biodiversity indicators to meet marine conservation policy and management demands is developing rapidly in the management realm, with a lag before academic publication detailing indicator development. Making best use of biodiversity indicators depends on sharing and synthesizing cutting-edge knowledge and experience. Using lessons learned from the application of biodiversity indicators in policy and management from around the globe, we define the concept of 'biodiversity indicators,' explore barriers to their use and potential solutions, and outline strategies for their effective communication to decision-makers.

Keywords: ecosystem approach, assessment, monitoring, baselines, thresholds, policy communication, cumulative effects

INTRODUCTION

Threats to marine biodiversity, from human activities such as fishing, shipping, coastal development, and energy production and from indirect pressures, like climate change, are increasing (Halpern et al., 2015), with only 13% of the world ocean still considered unimpacted by humans, or 'wild' (Jones et al., 2018). The loss of marine biodiversity impacts the resilience of ecosystems and the ability to maintain essential ecosystem services that support human life, such as food provision and water quality maintenance (Worm et al., 2006). The vulnerable state of global marine ecosystems and the need to sustainably monitor, assess, and manage habitats and species is increasingly recognized (Addison et al., 2017). Consequently, the assessment of the state of marine biodiversity, with associated biodiversity management and conservation measures, is now explicitly articulated in national (Department of Environmental Affairs and Tourism, 2004; Natural Resource Management Ministerial Council, 2010; Defra, 2018), regional (Cartagena Convention, 1983; European Commission, 2008b, 2011), and international (United Nations, 2010; United Nations General Assembly, 2015) legislative mechanisms. These mechanisms address both marine policy (the setting of regulation through legislation) and management (implementation of management plans, monitoring, evaluation and reporting on the status of the marine environment).

'Biodiversity' is "the variability among living organisms, from all sources, including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" [Convention on Biological Diversity (CBD); United Nations, 1992]. In other words, 'biodiversity' refers broadly to all species and habitats in an ecosystem, rather than simply the number of taxa. This definition is broad, encompassing all marine and coastal species and habitats. It is impossible to monitor and assess the state of all aspects of marine biodiversity, so the complexity of biodiversity is typically reduced in dimension by using indicators to summarize its key aspects. Indicators are therefore frequently used in marine policy and management to assess and communicate change in ecosystem state. They are the primary tool for assessing progress toward the CBD Aichi targets, which aim to halt global biodiversity decline (Balmford et al., 2005; Tittensor et al., 2014; United Nations General Assembly, 2015). Indicators as a concept have been used for decades in marine fisheries management [e.g., commercial fish stock management in South Africa and Europe (Plagányi et al., 2007; ICES, 2018), ecosystem-based fisheries management in Australia, New Zealand, the United States, and Canada (Sainsbury et al., 2000; Link et al., 2002; Methratta and Link, 2006; Fu et al., 2015)], in marine pollution regulation [e.g., assessment and management of marine sediment pollution in the North Sea (OSPAR, 2017k), and pollution assessment of fish, crustaceans, and molluscs in the Baltic Sea (HELCOM, 2018)].

Unlike more established indicators in marine fisheries and pollution regulation, which are measurable against a clear objective or target, techniques to develop indicators and targets and to assess the status of marine biodiversity to inform biodiversity management more widely, however, are new but rapidly developing (e.g., Tam et al., 2017). In Europe, for example, the Marine Strategy Framework Directive (MSFD) uses biodiversity indicators to assess the state of marine habitats and species, with the overarching objective of achieving 'Good Environmental Status' (GES) (European Commission, 2008b). Similarly in South Africa, the National Biodiversity Strategy and Action Plan aims to achieve 'Good Ecological Condition' which refers to ecosystems that are intact or largely intact with minimal modification from a natural condition (Department of Environmental Affairs, 2015). In the United States, implementing the ecosystem-based approach to management has moved to the forefront of efforts, including the development of quantitative indicators and criteria that can be used to assess overall ecosystem status (Leslie and McLeod, 2007). Where ecological data are lacking, such as in South Africa, expert judgment is often used to set targets for marine biodiversity indicators (e.g., Driver et al., 2011; Department of Environmental Affairs, 2015). Under the MSFD, while some biodiversity indicators already have agreed quantitative targets for individual regions (Defra, 2012; HELCOM, 2018), targets for other regions or indicators are still in development. Approaches to indicator development and target setting for effective management require not only a clear understanding of the system in question, which might need substantial amounts of data in some cases, but also explicit policy goals or objectives. These attributes may inhibit indicator development and policy uptake.

In June 2018, international developers and users of marine biodiversity indicators participated in a symposium and focus group entitled "From science to evidence – innovative uses of biodiversity indicators for effective marine policy and conservation" as part of the 5th International Marine Conservation Congress (IMCC5) in Kuching, Malaysia. The mission of the symposium and focus group was to form a community of practice for both users and developers of biodiversity indicators for marine policy and conservation, and to provide a forum to share successes and failures in developing and applying these indicators. Themes emerged which are common across geographic regions and political scales. This paper uses lessons learned from the application of biodiversity indicators in policy from around the globe to define the concept of biodiversity indicators, explore and discuss barriers and solutions to their use, and outline strategies for their effective communication to policy-makers.

CONCEPT, USE, AND SUITABILITY OF BIODIVERSITY INDICATORS

The wide definition of the terms 'indicator' and 'biodiversity,' as well as their broad applicability, can lead to confusion regarding the function of a biodiversity indicator. For instance, indicators can be defined simply as a "quantitative or qualitative variable that provides reliable means to measure a particular phenomenon or attribute" (USAID, 2009) or, using a process-oriented definition, as a "quantitative or qualitative factor or variable that provides a simple and reliable means to measure

achievement, to reflect changes connected to an intervention, or to help assess the performance of a development actor" (OECD, 2002). In a marine context, indicators have been defined as a tool "to monitor and assess the state of the marine environment and to manage human activities having an impact upon it" (European Commission, 2008b). Under the Convention of Biological Diversity (CBD), indicators are defined as tools "for assessing progress toward, and communicating the 2010s target at the global level" (United Nations Environment Programme, 2004), which hereby further extends their application and allows a broader use of terminology.

A bibliographic analysis of >2500 abstracts queried from the Web of Science database revealed a difference in treatment of the term 'biodiversity indicator' between academic scientists, marine policy-makers and managers (**Figure 1**). In publications on marine systems, 'ecosystem indicator' is used more commonly and synonymously with 'biodiversity indicator,' though the use of the 'biodiversity indicator' is increasing (see **Figure 1A**). Overall, we found that depending on the purpose, region, or policy context, indicator terminologies can differ despite representing similar ecosystem/biodiversity components. Nevertheless, biodiversity indicators are still often represented by conventional diversity indices such as species richness or evenness. These indices can be highly useful for summarizing and assessing community structures such as biogenic reefs or infaunal communities and linking them to anthropogenic pressures such as trawling (Cook et al., 2013; Fariñas-Franco et al., 2014; Van Loon et al., 2018). To provide sufficient information on ecosystem dynamics and processes for sound policy and management, however, other components such as biological trait diversity and ecosystem functioning can be similarly useful (Diaz and Cabido, 2001; Juan et al., 2007; Bremner, 2008; Pacheco et al., 2011).

The implementation of regional and international legislative frameworks has triggered a big rise in developing biodiversity indicators to determine the state of the ecosystem and its components in the last two decades. Publications on 'ecological,' 'ecosystem,' or 'biodiversity' indicators started to increase in the early 1990s after the United Nations Conference on Environment and Development with the resulting ratification of the CBD (**Figure 1A**) (United Nations, 1992) and the publication of the Organization for Economic Co-operation and Development (OECD) core set of indicators for environmental performance reviews (OECD, 1993). Publications addressing marine systems, however, started much later, in the mid-2000s, and so represent only 18% of all articles on biodiversity indicators, covering predominantly the temperate northern Atlantic ecoregion (see **Figure 1B**).

While the term 'biodiversity' may refer strictly to the diversity of biological components in an ecosystem, 'biodiversity' is increasingly used to reflect a much broader ecosystem view. This broader definition includes trophic interactions, network structure and system stability or resilience (e.g., Samhouri et al., 2009; Dakos et al., 2011), which is in line with the Convention on Biodiversity's definition of 'biodiversity,' above, and is often used by applied scientists, policy-makers, and managers. It is this second definition of 'biodiversity' that is used throughout this paper, due to its frequency of use in conservation. While we do not want to ignite a discussion on terminology superiority, we want to highlight the importance of understanding biodiversity in a wider context and propose a more flexible approach to the term 'biodiversity indicator' that includes multiple concepts such as

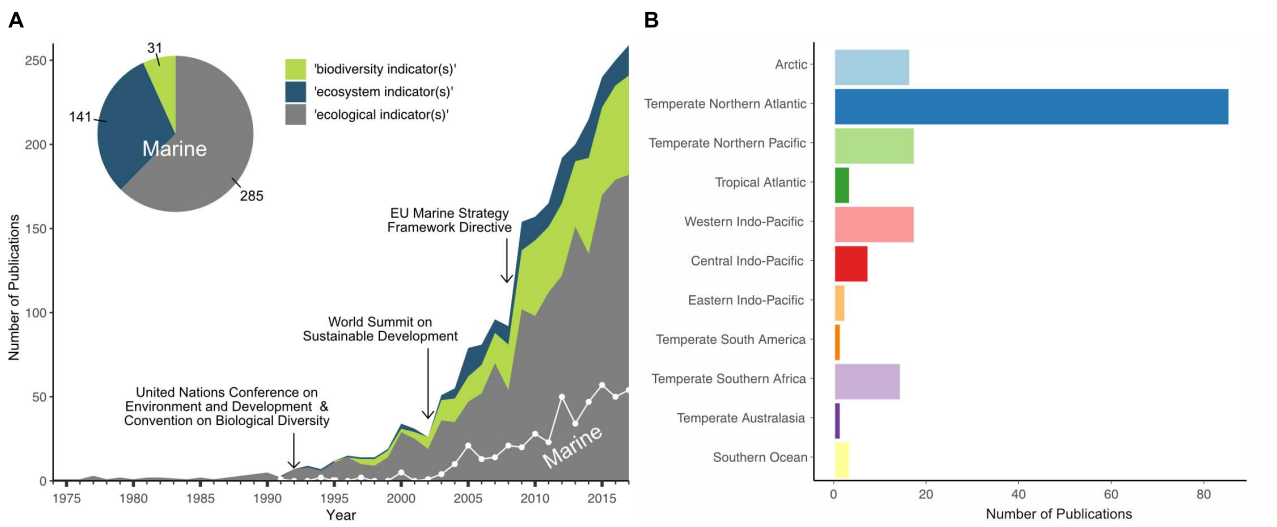


FIGURE 1 | Bibliographic analysis of publications on biodiversity, ecological, or ecosystem indicators in general and for marine systems specifically. **(A)** The number of publications using one of the indicator terms [biodiversity (green shading), ecosystem (blue shading), or ecological (gray shading) indicator(s)] between 1975 and 2017 (total of 2502), and the number of publications using these terms in relation to marine systems only (white trend line; total of 457), shown in relation to the years when three significant international or regional legislative frameworks were implemented. **(B)** The geographic distribution of a subset of 1430 publications across marine ecoregions (Spalding et al., 2007), extracted from publication abstracts and keywords. The bibliographic data were queried from the Web of Science database (accessed last Sept 18th, 2018).

TABLE 1 | Applications of biodiversity indicators relevant to marine environments and global marine conservation policy and management.

Indicators used for assessments	Examples of application	Spatial scale of application (presented in order of cited publications)
Status of, or changes in, species, habitats, or ecosystems	Beaugrand, 2005; Rochet et al., 2005; Blanchard et al., 2010; Shin et al., 2010; Shephard et al., 2014; Probst and Stelzenmüller, 2015	North Atlantic Ocean; France; Global; Global; Celtic Seas and Greater North Sea; North Sea
Track and communicate trends in quantity and quality of ecosystem services	van Oudenhoven et al., 2018	European seas
Signals prior to or after trending or oscillating changes	Lindegren et al., 2012; Cline et al., 2014	Baltic Sea; Global (lakes);
Impact of an anthropogenic pressure on the ecosystem	Shannon et al., 2010; Henriques et al., 2014; Coll et al., 2016	Global; Portugal; Global
Ecosystem stability or resilience	Samhouri et al., 2009; Vasilakopoulos et al., 2017	Global; Mediterranean Sea
Oceans at different spatial scales	Blanchard et al., 2010; Halpern et al., 2012; Coll et al., 2016; Uusitalo et al., 2016; Torres et al., 2017	Global; global; global; regional (European); single ecosystem (Baltic Sea)
Ocean biological indicators at different organizational levels (single species, individual guilds, entire food webs and trophic interactions)	Teixeira et al., 2016; McQuatters-Gollop et al., 2017	Global with European focus; European

Citations preceded by "e.g.," reflect one example of many.

ecosystem structure and functioning (as outlined by the Essential Biodiversity Variables for policy; Pereira et al., 2013).

In recent decades, a variety of approaches for the use of indicators in the marine environment have emerged, particularly in the temperate northern Atlantic ecoregion, which is largely triggered by the implementation of regional and international legislative frameworks (**Figure 1**). **Table 1** illustrates some examples of the applied versatility of biodiversity indicators, providing a wide-range of evidence types, at different ecological and spatial scales, for the assessment and management of marine biodiversity within the context of the policy questions they aim to address.

Despite the wide range of applications of biodiversity indicators observed during recent decades, specific selection criteria have been commonly accepted within the scientific community to determine indicator suitability for operational use. These include measurability, scientific basis, interpretability, and ease of communication, but also sensitivity and responsiveness to environmental changes, specificity, robustness with well-known pressure-state relationships, and links to identified targets and thresholds (e.g., OECD, 1993; FAO, 1997; Rice and Rochet, 2005; Heink and Kowarik, 2010; Kershner et al., 2011; Queirós et al., 2016; Otto et al., 2018a). Biodiversity indicators that address policy and management goals are likely to be most effective if the relevant stakeholders and decision-makers also perceive them to be credible, salient, and legitimate (Cash et al., 2003; van Oudenhoven et al., 2018). Linking indicators to environmental conditions and ideally to management measures requires a good understanding of indicator responses to pressures and a sound testing of indicator performance, which is often lacking for biodiversity indicators (Rossberg et al., 2017). Thus, new modeling approaches and decision support tools are emerging to tackle the performance evaluation of indicators for assessing the health status of marine ecosystem and biodiversity components (Hayes et al., 2015; Lynam et al., 2016; Otto et al., 2018a; Shin et al., 2018) (see also section *Linking biodiversity indicators to ecosystem change*). To complement assessments of state, additional pressure indicators can be useful, particularly to measure the impacts of human activities on the system when there can be a long time-lag before natural processes can be expected to respond (Rossberg et al., 2017).

Indicators that lack a clear link to a defined pressure, however, can still contribute effectively to the assessment and management of biodiversity. These indicators without clear links to defined pressures, known as 'surveillance indicators' (Shephard et al., 2015), may not be able to be assessed against quantitative thresholds, but can still provide contextual information on either wider ecosystem impacts of pressures or underlying environmental change (Bedford et al., 2018). Critically, indicators used in a 'surveillance' context should still increase the knowledge base from which to make management decisions. For example, a suite of 'Essential Ocean Variables' for biodiversity and ecosystem change has been identified by Miloslavich et al. (2018) to effectively reduce the complexity of ecosystem processes for a summary of ecosystem state. Although not linked to specific defined pressures, the impacts of both direct anthropogenic pressures and climate change on these ecosystem processes can be monitored and assessed, providing holistic surveillance information to support management.

BIODIVERSITY INDICATORS IN POLICY AND MANAGEMENT: NEEDS, BARRIERS, AND SOLUTIONS

Indicator development is challenged by the need to establish associated targets, political acceptance, and evaluation of confidence to support widespread use for management of biodiversity (**Table 2**).

Biodiversity Indicators Linked to Policy and Management

Often, scientists develop biodiversity indicators in academia, usually to address a scientific problem but also to assess the ecosystem status within the context of specific policies, and then publish their results in the scientific literature. A recent review by Bal et al. (2018) showed that indicators (in this case, those

TABLE 2 | Needs, barriers, and solutions to the development and use of marine biodiversity indicators.

Need	Barrier	Solution
Biodiversity indicators linked to policy and management.	Siloed development of indicators, resulting in indicators that do not meet the needs of decision-makers.	Co-production of indicators by scientists and decision-makers (Lemos and Morehouse, 2005).
Appropriate biodiversity data are required to inform indicators.	Insufficient data to capture spatial and temporal variability of marine ecosystems due to: - High costs of data collection. - Vast scales (spatial and temporal) over which ecological processes and patterns occur. - Non-policy oriented focus of historic data collection. - Lack of capacity for marine management infrastructure.	Pragmatic approach to indicator design that supports the combination and repurposing of existing data sets (OSPAR, 2017a,b,g,h). Risk-based approach to target intensive monitoring in order to answer specific and clear policy question (Elliott et al., 2018; Turrell, 2018). Use of earth observation and models to supplement *in situ* data (Elith et al., 2006; Butchart et al., 2010; Bean et al., 2017; Strong and Elliott, 2017; Pettorelli et al., 2018). Use of human impact (pressure) data where biodiversity monitoring data are unavailable (Halpern et al., 2012). Use of citizen science programs for data collection (Hodgson, 2000; Goffredo et al., 2010; Gillett et al., 2012; Bull et al., 2013; ICES, 2017; Stuart-Smith et al., 2017).
Linking biodiversity indicators to ecosystem change.	Biodiversity indicators may respond to multiple pressures, including climate change, making it difficult to identify causes of change. Systems may respond non-linearly to pressures, obscuring indicator interpretation.	Integration of biodiversity indicators during assessments increases confidence in identify causes of change (Smith et al., 2016). Ecosystem modeling to identify the important pressure-state pathways (Fulton et al., 2005; Lynam et al., 2016; Shin et al., 2018). A range of modeling tools can examine non-linear indicator-pressure relationships (e.g., Hyder et al., 2015; Otto et al., 2018a,b).
Using biodiversity indicators to measure progress toward policy goals	Setting targets for biodiversity indicators is challenging due to: - Difficulty in identifying reference conditions. Political resistance to targets.	Reference conditions can be constructed based on spatial or time-series data or using models (Borja and Tunberg, 2011; Borja et al., 2012; OSPAR, 2017c,f,i,j; Rossberg et al., 2017) allowing targets to be set at an acceptable distance from the reference conditions. Trend based approaches do not require indicators to reach a specified endpoint or target point (Butchart et al., 2010). Close science-policy collaboration can produce evidence-based SMART targets (Heritier, 2002; Cvitanovic and Hobday, 2018). Decision triggers may be used instead of targets to trigger management action (Addison et al., 2016).

based on species traits) developed in academia and reported in the scientific literature typically fail to address decision-making requirements for biodiversity management, with only 21% of studies detailing how indicators explicitly address policy objectives. This review clearly demonstrates the broad use of the term 'indicator,' but it also shows that the academic approach to indicator development is often driven by scientific questions rather than a response to policy needs, or if policy-focused takes place outside the policy process. In such cases indicators are frequently not formally incorporated into the assessment of management objectives and targets (Bal et al., 2018). Regardless of the scientific soundness of an indicator, or even the appropriateness for a specific policy, the lack of involvement of end-users (e.g., marine managers, policy-makers, and stakeholders) during the development of indicators may result in unsuccessful implementation of the outputs or even the application and use of the indicator itself.

A solution resulting in fit-for-purpose biodiversity indicators is to co-produce indicators, with scientists providing the scientific input and decision-makers providing the policy steer (Lemos and Morehouse, 2005; Hayes et al., 2015; Bolman et al., 2018; Cvitanovic and Hobday, 2018; De Juan et al., 2018). Co-production spans the science-policy interface and is an iterative process, with each party relying on the other's experience and expertise to gain a deeper understanding of the current science and policy landscapes, opportunities, and limitations (Lemos and Morehouse, 2005). The co-production of biodiversity indicators has resulted in their successful use in marine policy and management (e.g., in Australia and Europe; Pocklington et al., 2012; OSPAR, 2017d). For

example, biodiversity indicators developed for the 2017 OSPAR Intermediate Assessment followed this process (OSPAR, 2017d). The indicators were developed by scientists with significant and consistent input from policy-makers to ensure the indicators fulfill policy obligations. As a result, the regional biodiversity assessments can be used by EU member states for the fulfillment of the MSFD (OSPAR, 2017d).

Data Requirements for Biodiversity Indicators

A basic requirement when developing a biodiversity indicator is an understanding of the types of data available and a critical evaluation of the temporal and spatial scales that are appropriate for the ecological processes being assessed and the pressures on the marine ecosystem. Large-scale monitoring programs collecting time-series data are very rare, particularly in offshore areas, mainly due to the costs of data collection (Koslow and Couture, 2013). Marine monitoring needs to be well governed, cost-effective, organized, transparent, open, designed on a scientific basis, and "fit for purpose" (Turrell, 2018). Furthermore, data collection for biodiversity indicators ideally should be tailored to the policy questions the indicator is trying to address, for example by developing relevant sampling strategies and power analyses to establish the level of sampling effort required to detect community change at a particular scale.

However, data-intensive indicators, even if they are high in confidence and accuracy, are not always practical for large scale biodiversity assessments, such as required for management of regional marine environments, especially for those ecosystem components for which monitoring is expensive. This lack of practicality is a particular challenge for evaluating ecological processes or distributional patterns of habitats or species which require monitoring surveys over a large spatial area as compared to verifying the presence of, for example, a sensitive species in an MPA (Barrio Froján, 2016).

The costs of data collection can pose a barrier to indicator development, particularly for low income countries, which contain some of the world's most diverse species and habitats (Tittensor et al., 2010; Ramírez et al., 2017), but are generally poorly monitored due to economic challenges and lack of infrastructure and scientific experts (Danielsen et al., 2000). While high-income countries tend to pose more threats to marine ecosystems (Beck et al., 2011; Thurstan et al., 2013; Halpern et al., 2015; Fariñas-Franco et al., 2018), a lack of fundamental biodiversity research, capacity and coordination of information in low-income countries makes them highly vulnerable, particularly to climate change (Bellard et al., 2014). Many marine and coastal ecosystems are highly diverse, yet there is a lack of fundamental biodiversity research required to understand processes and species distributions in the marine environment (Griffiths et al., 2010). This lack of investment also extends to the capacity and coordination of marine biodiversity information within and outside of the scientific community which can prevent its use within decision-making (Atkinson et al., 2016).

A solution to overcome data shortages or limitations to access, involves a pragmatic approach to indicator construction, together with good use of existing ecological datasets for the relatively new purpose of informing biodiversity indicators for policy and management. Data limitations often can be overcome by constructing indicators with the flexibility to use data from multiple sources (e.g., OSPAR, 2017a,b,g,h) or by using a risk based approach to identify areas where targeted, more intensive monitoring should be concentrated (Elliott et al., 2018).

Additional solutions include setting clear monitoring objectives and clearly articulating the decision context that defines the temporal and spatial requirements for management decisions. This will ensure that the data required to inform biodiversity indicators are collected in a cost efficient manner (Turrell, 2018). In cases where extensive monitoring data are needed but not practical to collect, the use of alternative data sources, such as Earth observation, rather than data solely collected via *in situ* monitoring, can facilitate regional biodiversity assessments (Bean et al., 2017; Strong and Elliott, 2017; Pettorelli et al., 2018). For example, models combining physical, geological and biological parameters are currently being used to evaluate the extent and distribution of benthic habitat types at regional scale (OSPAR, 2017b). Furthermore, modeled species distributions can provide data to develop indicators such as the presence/absence of species and biotopes based on their environmental preferences for areas where survey data are missing or limited in extent (Elith et al., 2006; Butchart et al., 2010). They can also help in identifying impact hot spots and evaluating management actions (Guisan et al., 2013).

South African practice presents a possible solution to the challenges of monitoring marine biodiversity (Atkinson et al., 2016). Broad scale assessments of the state of South African marine ecosystems have been based on the Ocean Health Index method (Halpern et al., 2008, 2009) which uses cumulative human impacts in the absence of spatially extensive biodiversity monitoring data. This method can enable low income countries and other regions with limited biodiversity data to arrive at an indicative national scale assessment of biodiversity. The Ocean Health Index assumes that areas of high human pressure are in poor ecological condition. While useful, the method may not capture fine-scale natural variability, and can fail to identify areas of high resilience as well as the presence of unique or vulnerable ecosystems. Nevertheless, South African policy-makers have so far accepted this method of assessment, acknowledging the challenges and limitations to assessing the condition of the marine environment for the entire exclusive economic zone of South Africa using impact, or pressure, information in the absence of biodiversity data (Driver et al., 2011; Department of Environmental Affairs, 2015). To evaluate the outcomes of this practice, these methods should be verified with empirical evidence at varying scales using ecological monitoring data where available (Sink et al., 2012).

Involving the public in monitoring may be another cost-effective solution to the labor-intensive data collection required to inform biodiversity indicators (Thiel et al., 2014; Freiwald et al., 2018). Limitations on data collection are common, such as lack of standardization and spatio-temporal coverage, particularly in

geographical areas which are greatly impacted but less accessible to the public. Despite these challenges, there are some notable regional and global citizen science programs that are increasing data coverage for some aspects of the marine environment for use in policy and management such as: Seasearch, which uses volunteer scuba divers to collect species data around the coast of Britain and Ireland[1]; Reef Check and Reef Life Survey, which are global programs that monitor the health of temperate and tropical reefs (Hodgson, 2000; Stuart-Smith et al., 2017); public monitoring of European seabirds (ICES, 2017); and a series of national citizen science programs for temperate rocky reefs in California (Gillett et al., 2012), subtidal habitats in the United Kingdom (Bull et al., 2013), and marine biodiversity health in northern Italy (Goffredo et al., 2010).

Linking Biodiversity Indicators to Ecosystem Change

Developing biodiversity indicators that are responsive to a defined anthropogenic pressure or linking biodiversity indicator change to a single manageable pressure is often desired by policy-makers but is scientifically challenging to achieve. Micheli et al. (2013) found that ∼60–99% of the territorial waters of EU member states were heavily impacted as a result of multiple pressures, rather than one individual stressor. These multiple pressures, which include climate change, can have cumulative and synergistic effects on biodiversity components, reflected by indicator state (Côté et al., 2016). For example, warming temperatures have been shown to interact with fishing pressure on temperate fish stocks (Kirby et al., 2009) and with multiple stressors including pathogens on coral reef ecosystems (Ban et al., 2014). Furthermore, biodiversity components are fundamentally linked through trophic interactions, affecting biodiversity indicators. Torres et al. (2017) showed that no pressure-state relationships for fish indicators in the Central Baltic Sea could be found unless predator-prey feedback or density dependence was accounted for. These complex and interacting drivers obscure the interpretation of change in biodiversity indicators. For example, the limited understanding of the effects of environmental drivers on the variation of Porifera and Anthozoa assemblages across the North of Scotland and Celtic Sea is hindering the ability to accurately measure ecological responses of benthic rocky reef indicators to direct anthropogenic pressures (Haynes et al., 2014).

Multiple biodiversity indicators may respond to the same anthropogenic pressure. Integrating information from a range of biodiversity indicators is a solution that can help to provide an overall assessment of the ecosystem (Elliott et al., 2018) and clarify the main drivers of change affecting a system (Smith et al., 2016). Although significant development is often required, ecosystem modeling can provide a comprehensive means to detect change in multiple biodiversity components and identify the important pathways by which impacts from pressures can cascade through an ecosystem (Lynam et al., 2016). Thus embedding indicators within a model framework can demonstrate key pressure-state linkages (Fulton et al., 2005; Shin et al., 2018), although it must be noted that data quality may impact model performance. Such models can then be used to examine the effects on biodiversity indicators of potential management measures or climate change through scenario testing (e.g., Mackinson et al., 2018; Queirós et al., 2018).

Another factor to consider when linking indicators to pressures is the non-linearity in marine ecological systems. For some marine ecosystems abrupt community shifts have been reported (e.g., Hare and Mantua, 2000; Frank et al., 2005) that can only be explained by non-linear state responses to abrupt changes in pressures (Scheffer and Carpenter, 2003). Non-stationarity, i.e., spatio-temporal change in the state-pressure relationship (Hunsicker et al., 2016), impedes the development of robust indicators that behave in a consistent and predictable way. A new tool, the R package 'INDperform' (Otto et al., 2018b) accounts for these dynamics and allows the user to explicitly test for non-linear and non-additive indicator-pressure relationships. The package builds on a quantitative framework for selecting and validating the performance of indicators tailored to specific management needs (Otto et al., 2018a) and offers additional functions to quantify the robustness of these models, identify temporal indicator changes, test for indicator redundancy, and visualize performances. While single indicator-pressure models, such as offered in INDperform, can easily be applied to any number of indicators and pressures they cannot account for synergistic or counteracting effects of multiple pressures or estimate trade-offs between individual indicators. For this, more complex modeling tools are required, which in turn can be difficult to communicate, may require many assumptions, and take longer to build (Hyder et al., 2015).

Using Biodiversity Indicators to Measure Progress Toward Policy Goals

Policy goals are often definitive, moving beyond broad-scale visions, and instead specifying a target condition that needs to be reached to meet the goal. An example of this is "...the abundance/extent, distribution and condition of marine species and habitats are in line with prevailing environmental conditions" from Descriptor 1 Biological Diversity of the EU's Marine Strategy Framework Directive (2008/56/EC). Such an approach has long been used to assess indicators of environmental quality, including concentration of contaminants in water bodies (e.g., mercury, PCBs, nitrates) and of harmful gasses in the air (e.g., carbon monoxide, sulfur dioxide). For these indicators, laboratory tests establish safe limits which can then be used to define desirable target levels for environmental conditions (European Commission, 2008a). Setting quantitative targets that define a good or favorable condition for biodiversity indicators, however, is much more challenging, as our understanding of ecological processes influencing the recovery of species or habitats and the associated ecosystems functions is more limited. Consequently, many biodiversity indicators currently still lack associated defined targets (Teixeira et al., 2016).

The most common first step to defining targets for biodiversity indicators is to establish a baseline against which future change

[1] http://seasearch.org.uk/

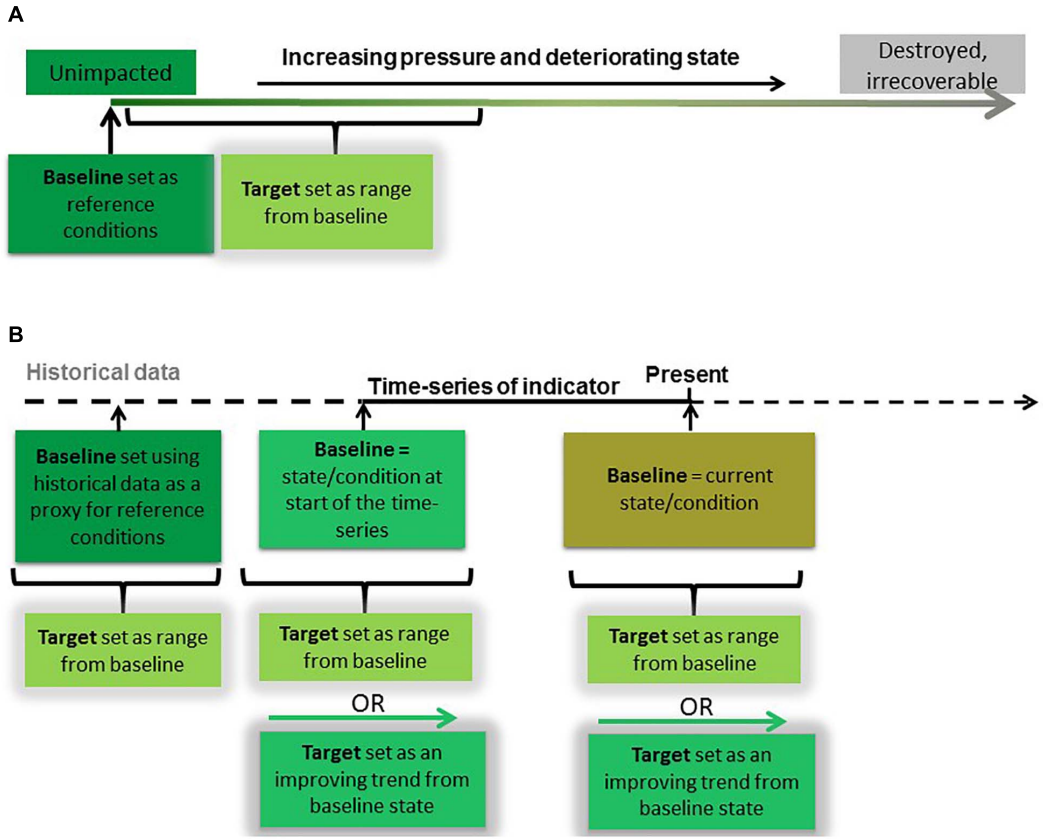


FIGURE 2 | Establishing baselines and setting targets under two scenarios of biodiversity data availability. **(A)** The relative condition of the indicator is known, with data available representing unimpacted conditions (reference conditions). In this case, an indicator target can be set as a range of indicator values within a specified distance from the baseline reference conditions. **(B)** The relative condition of the indicator is not known, and no data representing reference conditions are available. In this case, time-series data are used to establish baseline conditions and set targets. Baselines can be set using (1) historical data, such as from an alternative data source or model, (2) the earliest time-series data available, or (3) data representing current conditions. Targets can then be set as a range or as an 'improving' trend from baseline state.

in condition can be measured (**Figure 2**). The most robust approach to baseline setting is to first establish a 'reference condition' (Borja et al., 2012; Greenstreet et al., 2012; OSPAR, 2012; Probst et al., 2013) or "natural range" (Rossberg et al., 2017) which will enable the full effects and changes caused by anthropogenic pressures to be evaluated (Van Loon et al., 2018). Reference conditions can be derived from information on species and habitats from areas where human pressure is considered negligible or non-existent but that information must be shown to be applicable to other areas (Borja and Tunberg, 2011). Reference conditions for marine biodiversity indicators, however, can be difficult to identify as areas of the marine environment that have been unimpacted by human pressures are increasingly scarce (Jones et al., 2018). Furthermore, time-series for most indicators are not long enough to include a time when human impacts were absent or negligible (Butchart et al., 2010; Dornelas et al., 2018). Unimpacted conditions are particularly difficult to identify for mobile species such as birds, marine mammals, fish and turtles because they move between impacted and unimpacted areas (OSPAR, 2012). Modeling, however, can be used to predict reference conditions, based on knowledge of human pressures and their impact on the state of the indicator (Borja et al., 2012; Rossberg et al., 2017). Once reference conditions are established, targets can then be set that are within a specified distance from them (OSPAR, 2012), where the acceptable target range for this distance is dependent on the rate of recovery of the state in question (Rossberg et al., 2017).

In the absence of empirical or modeled reference conditions, recent assessments of birds, seals, and fish in the NE Atlantic have used the start of time-series to define baselines for indicators (**Figure 2**) (OSPAR, 2017c,f,i,j). The risk with this approach is that the baseline is set at a value that represents a degraded condition which may or may not be within the acceptable target range of the ecosystem state. If targets are then set close to the baseline condition, this may jeopardize any improvement or recovery beyond that observed recently. This concept is referred to as Shifting Baseline Syndrome (Pauly, 1995; Pinnegar and Engelhard, 2008; Papworth et al., 2009) and can result in targets lacking in ambition (Plumeridge and Roberts, 2017) or worse, 'locking in loss' (Maron et al., 2015). Objective baselines and targets can be set once we improve our understanding of pressure-state relationships and the influence of the environment

on them. Duarte et al. (2009) caution that it might not be possible for an indicator to return to a historic state because of fundamental alterations to the ecosystem caused by long-term or chronic effects of pressures or similarly changes in environmental conditions (Möllmann et al., 2009). In such cases, baselines that denote reference conditions would need to be set at a theoretical natural state, which could be achieved in the future if all current human impacts were removed (Rossberg et al., 2017). If the policy goal is sustainable use, the indicator targets should allow components of the ecosystem to achieve the theoretical natural state in a societally acceptable period of time (such as within a human generation) if all current human activities were to cease (Rossberg et al., 2017). To ensure the highest probability of such a recovery, impacts by human activities on structure, productivity, function and biological diversity of the ecosystem should be minimized (Garcia et al., 2003).

Where indicators are required to measure progress toward broad-level policy goals and visions, trend-based targets provide an appropriate solution. Trend-based assessment approaches are relatively simple to apply and communicate and are useful to inform on the progress of management in helping to recover degraded habitats or ecosystems or depleted species populations. For example, the Convention on Biological Diversity Aichi Target 12 is a broad-level vision stating that "By 2020 the extinction of known threatened species has been prevented and their conservation status, particularly of those most in decline, has been improved and sustained" and is used to assess progress toward Strategic Goal C "to improve the status of biodiversity by safeguarding ecosystems, species and genetic diversity" (United Nations, 2010). Measuring progress toward this goal, however, does not require indicators to reach a specified endpoint or target point, but instead assessment is based on indicator trend.

An additional barrier to setting targets for biodiversity indicators is that political resistance can be generated by a lack of agreement on the level of ambition by different parties, for example, across different countries sharing the same sea area. This can stem from a lack of understanding of what the indicator values signify and/or uncertainty around the implications or consequences of missing a target. Failure to meet targets may carry reputational risks or could lead to costly remedial measures such as changes in regulation or management, which may create resistance to targets from industry. Some of these political sensitivities can be alleviated through scientists working closely with policy leads to co-produce SMART targets that make the most of the available evidence (Cvitanovic and Hobday, 2018). For international targets, fora involving national representatives from science and policy can help to achieve international consensus and ensure targets are adopted by countries rather than imposed upon them (Heritier, 2002; OSPAR, 2017c,f,i,j).

Decision triggers are less contentious than firm targets and can provide a useful link from monitoring data to management decisions. Decision triggers are becoming an appealing tool for conservation managers to help support decision-making by providing clarity about when and how to act; improving transparency of organizational decisions; removing the need for guess work; guarding against the paralyzing effects of uncertainty; and preventing negative conservation outcomes (Addison et al., 2016). Decision triggers represent a point or zone in the status of a monitored variable indicating when management intervention is required to address undesirable ecosystem changes (Cook et al., 2016). Decision triggers can be set using a number of methods, depending on the availability of scientific data and expertise, the number of objectives for management and the resources available (Bie et al., 2018).

STRATEGIES FOR COMMUNICATING BIODIVERSITY INDICATORS TO POLICY

Effective communication of biodiversity indicators and assessments is integral to their uptake by policy-makers and managers. Critically, the target audience must be identified so indicator communication can be tailored appropriately. The group 'policy-makers' is often used as a generic term for decision-makers at multiple levels, including local councilors, environmental managers, civil servants, congress people, Members of Parliament (MPs), and ministers, among others. These subgroups use biodiversity indicators in different ways to make decisions and therefore require information in different formats with varying levels of associated detail and specificity.

Regardless of the audience, biodiversity indicator communication must be clear, transparent and easy to understand to support their legitimate use in decision-making. There are different ways to present indicator results and assessments, each of which involves trade-offs between the complexity of biodiversity information and the simplicity of the product required for clear communication (**Figure 3**). The simplest methods of indicator communication use traffic lights summaries (United Kingdom Marine Monitoring and Assessment Strategy, 2010; Driver et al., 2011; Karnauskas et al., 2017) or trend lines (WWF, 2016), which are simple visual illustrations of indicator change and are easily understood by non-scientists. These approaches often include composite indicators that are constructed by integrating numerous indicators to provide a single value (e.g., the Ocean Health Index, 2017) or trend (e.g., the Living Planet Index; WWF and ZSL, 2016). These products can deliver a simple but powerful, attention-grabbing message to a wide and diverse policy- and decision-making audience. However, the simplicity of these approaches, and lack of associated written narrative, also brings a risk that the audience may misinterpret the message conveyed by the indicator results. It is therefore the responsibility of scientists and managers to communicate results unambiguously, in a way that effectively takes account of any uncertainty in the results (Fischhoff and Davis, 2014).

Conversely, more complex communication methods such as summary report cards (e.g., Carey et al., 2017; European Environment Agency, 2017; Marine Climate Change Impacts Partnership, 2017) and narrative reports (e.g., Conservation of Arctic Flora and Fauna, 2017; Evans et al., 2017; OSPAR, 2017d) can provide a strong written narrative and contextual information, reducing the likelihood of

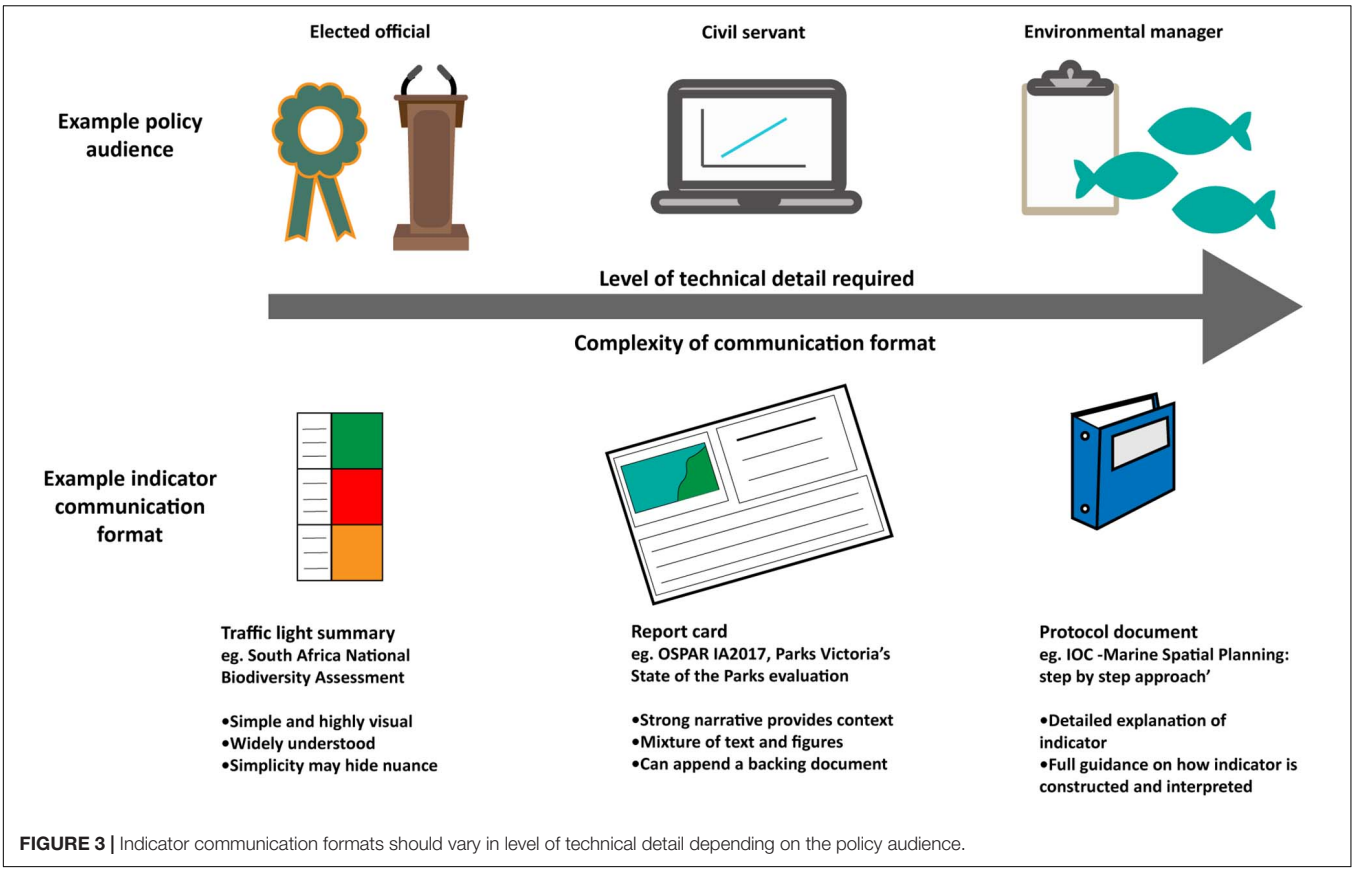


FIGURE 3 | Indicator communication formats should vary in level of technical detail depending on the policy audience.

misinterpretation by policy-makers. Protocol documents (e.g., Ehler and Douvere, 2009) are even more detailed, acting as a 'user guide' for indicators.

For all policy audiences, confidence in indicator assessments must also be clearly communicated. Addison et al. (2017) suggest that confidence in indicator assessments can be communicated through a variety of ways. For example, relatively simple categorical estimates of confidence in scientific robustness and/or supporting data informing indicator assessments can be applied. Some examples from Australia and Europe include reporting simple 'high, medium, and low' confidence designations (e.g., Carey et al., 2017; OSPAR, 2017e), measuring comparability with previous assessments [e.g., designating current indicator assessments as 'comparable,' 'somewhat comparable,' or 'not comparable' with previous assessments (e.g., Evans et al., 2017)], and making the evidence (data, metadata, reports, and papers) used in assessment transparent and accessible (e.g., Ocean Health Index, 2017; OSPAR, 2017d).

Progress toward achieving any associated targets may also be appropriate to communicate to policy-makers, including some measure of distance from the associated target as well as an indication of management interventions needed to achieve the target in the future (Andersen et al., 2014; HELCOM, 2018). Emphasizing socioeconomic needs linked to biodiversity indicators and assessment, such as ecosystem service provision, can help articulate policy relevance and increase usefulness of biodiversity indicators and assessments. Delivering the right indicator information in the right communication format for the right audience is therefore key to successful use of biodiversity indicators and assessments. For example, environmental managers who must make rapid management decisions require a higher level of detail about indicator implementation and interpretation than a national minister, who may only need to understand high-level information (**Figure 3**).

The co-development of indicators by scientists working closely with policy-makers can facilitate feedback on product communication format to ensure that the final indicators or assessment products are useful for policy-makers. Furthermore, indicator co-production allows the articulation of scientific confidence limits and risks, enabling agreement on a way to consider and express these limitations in assessments (Addison et al., 2017; Bolman et al., 2018). This is a critical, and often iterative, step in biodiversity indicator and assessment utility. Recent examples of this collaborative approach to indicator development are the OSPAR Intermediate Assessment of the Northeast Atlantic (OSPAR, 2017d) and the HELCOM Holistic Assessment of the Baltic Sea (HELCOM, 2018) where scientists worked closely with policy-makers to develop a suite of marine biodiversity indicators. The science-policy working groups co-developed communication products tailored to the requirements of two levels of decision-makers. Firstly, a detailed assessment report containing information about indicator development, assessment methods, and the interpretation of indicator results was developed for government

civil servants to use for reporting. Secondly, a two-page report card for elected officials, containing simple figures, provided a high-level overview of assessment results. Close working across the science-policy interface therefore resulted in biodiversity assessment products which meet the needs of both policy audiences.

Lastly, evidence-based decision making is essential for effective biodiversity management in the marine environment and in that sense promotes the use of user friendly mathematical or statistical models, such as decision-support tools that can translate science into policy (Pınarbaşı et al., 2017). Multifunctional decision support tools have been developed for a wide range of components in marine management, some of which may be useful to communicate results to decision-makers or to identify trade-offs and perform scenario analyses. These types of DSTs are particularly useful for detecting changes in marine ecosystems by performing scenario analyses on key drivers or biodiversity indicators within marine systems.

Although the scientific process in developing a set of indicators may be complex, the outputs should be simplified such that the outputs are connected to the human or social context in which they will be used. Technical DSTs or complex indicators may result in a disconnection between the objective of the indicator and its utilization in the decision-making process (Bolman et al., 2018). Therefore, simplifying complexity should rather focus on the communication of the scientific outputs rather than on the actual development of the indictors or tools. Communicating biodiversity indicators should include emphasizing key trends or sensitive parameters to communicate the dynamics within complex marine systems, in the format most useful to different decision-makers (e.g., decision support tools, report cards, or web-based interfaces).

CONCLUSION

As we enter the UN Decade of Ocean Science for Sustainable Development (UNESCO, 2018) a concerted effort will be required to develop strategies to meet the UN global goal to "Conserve and sustainably use the oceans, seas and marine resources for sustainable development" [Sustainable Development Goal 14 (United Nations General Assembly, 2015)]. Marine biodiversity indicators are likely to be critical to meeting the targets associated with this ambitious goal.

In the context of marine management, we highlight a holistic approach to understanding the term 'biodiversity indicator' to include ecosystem structure and functioning. Several challenges around biodiversity indicator development limit the widespread implementation in biodiversity management. Firstly, the policy application of marine biodiversity indicators varies across geographical regions and is currently most common in, but not limited to, high income countries with established monitoring programs. Where marine biodiversity indicators are in use for policy assessments, these indicators often use region-specific terminologies and data requirements, and were created for specific policy drivers. Additionally, marine ecosystems are complex, non-linear systems and links between internal interactions and exogenous pressures frequently distort human intuition of the marine system and hence management approaches. Marine management, and the development of biodiversity indicators to support management, thus require methods of analysis and decision-support tools that recognize multiple forms of complexity.

Formation of a community of practice was a key aim of this IMCC symposium and focus group, and these sessions revealed that the concept of biodiversity indicators is most useful when kept broad and flexible in both definition and application. A community of practice will facilitate knowledge exchange between indicator users to find alternative solutions for the common challenges outlined in this paper. Solutions to many of the challenges facing the policy application of marine biodiversity indicators were discussed and further developed and are now described in this paper. Some solutions require advanced numerical expertise while others address barriers by adopting innovative solutions involving citizen science data collection, combining multiple datasets to populate indicators, communicating assessment results in audience-specific formats, and enhancing collaborations within the international scientific community. The key to overcoming many barriers to biodiversity indicator uptake is to include policy-makers from the start of indicator development to ensure that implementation needs are met. It is our hope that the solutions outlined here will support the use of biodiversity indicators for marine policy, management, and conservation, helping us to meet the UN aspiration of the sustainable use of our oceans, seas, and marine resources.

DATA AVAILABILITY

All datasets generated for this study are included in the manuscript and/or the supplementary files.

AUTHOR CONTRIBUTIONS

AM-G, IM, CV-H, JB, SO, PA, and CL conceived and led the research and designed and led authorship of the manuscript. PG, EV, KS, DB, EM-W, and HN contributed to research and manuscript authorship.

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Key Challenges in Advancing an Ecosystem-Based Approach to Marine Spatial Planning Under Economic Growth Imperatives

Amanda T. Lombard[1], Rosemary A. Dorrington[2], Jodie Romay Reed[3], Kelly Ortega-Cisneros[1,4], Gwenith Susan Penry[5], Lorien Pichegru[6], Kaylee Pam Smit[3], Estee Ann Vermeulen[3], Minke Witteveen[3], Kerry J. Sink[1,7], Alistair M. McInnes[8] and Tayla Ginsburg[9]*

[1] Institute for Coastal and Marine Research, Nelson Mandela University, Port Elizabeth, South Africa, [2] Department of Biochemistry and Microbiology, Rhodes University, Grahamstown, South Africa, [3] Department of Oceanography, Institute for Coastal and Marine Research, Nelson Mandela University, Port Elizabeth, South Africa, [4] Department of Ichthyology and Fisheries Science, Rhodes University, Grahamstown, South Africa, [5] Department of Zoology, Institute for Coastal and Marine Research, Nelson Mandela University, Port Elizabeth, South Africa, [6] Department of Science and Technology/National Research Foundation Centre of Excellence at the Percy FitzPatrick Institute of African Ornithology, University of Cape Town, Cape Town, South Africa, [7] South African National Biodiversity Institute, Port Elizabeth, South Africa, [8] Department of Zoology, DST/NRF Centre of Excellence at the Percy FitzPatrick Institute, Nelson Mandela University, Port Elizabeth, South Africa, [9] Department of Zoology, Nelson Mandela University, Port Elizabeth, South Africa

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Edward Jeremy Hind-Ozan, Department for Environment, Food and Rural Affairs, United Kingdom

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Di Jin, Woods Hole Oceanographic Institution, United States

***Correspondence:**
Amanda T. Lombard
mandy.lombard@mandela.ac.za

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In 2017, South Africa became the first African country to draft Marine Spatial Planning (MSP) legislation. The underlying legal framework supports the achievement of ecological, social and economic objectives, but a national policy to fast track the oceans economy provides a challenge for ecosystem-based approaches to MSP. During the 2018 International Marine Conservation Congress, we convened a session to present particular challenges that will likely apply to any developing country seeking to increase profits from existing, or proposed, marine activities. Here we present six multi-disciplinary research projects that support ecosystem-based approaches to MSP in South Africa, by addressing the following knowledge gaps and specific key challenges: (1) the lack of data-derived measurements of ecosystem condition (and the need to validate commonly-used proxy measures); (2) the need to develop models to better understand the potential impacts of climate change on food webs and fisheries; (3) the slow implementation of an ecosystem approach to fisheries management, and the need to implement existing legal instruments that can support such an approach; (4) the paucity of evidence supporting dynamic ocean management strategies; (5) the requirement to manage conflicting objectives in growing marine tourism industries; and (6) the need to adopt systems thinking approaches to support integrated ocean management. We provide examples of specific research projects designed to address these challenges. The ultimate goal of this research is to advance a more integrated approach to ocean management in South Africa, using tools that can be applied in countries with similar socio-political and environmental contexts.

Keywords: ecosystem condition, oceans economy, climate change, dynamic ocean management, scenario planning, system dynamics models, complex systems, trade-offs

INTRODUCTION

South Africa's large exclusive economic zone includes the Indian, Atlantic, and Southern Oceans. Management of this ocean space has traditionally been undertaken within sectors, leading to conflict amongst sectors, and between commercial and environmental interests. As the demand for ocean space and marine resources increases, in response to an economic growth imperative defined by the National Planning Commission [NPC] (2012), a more integrated approach to management is required to ensure that both ecological and socio-economic objectives are met. Marine spatial planning (MSP) has emerged in many countries as the preferred process to achieve this integration, and in 2017, South Africa became the first African country to draft MSP legislation (Marine Spatial Planning Bill [B9-2017], 2017). In December 2018, the Bill was passed by both the South Africa National Assembly and National Council of Provinces and sent to the President to be signed into law. The underlying legal framework of the Bill supports the achievement of ecological, social and economic objectives, but a national policy to grow the oceans economy (Operation Phakisa)[1] provides a challenge for mainstreaming ecosystem-based approaches to MSP into policy and decision making. The adoption of ecosystem-based approaches to MSP is globally endorsed (Ehler, 2008; Gilliland and Laffoley, 2008; Halpern et al., 2008; Foley et al., 2010; Westholm, 2018; Kirkman et al., 2019), and embodied in the United Nations Sustainable Development Goals, particularly Goal 14 (United Nations, 2015), as well as the Nairobi Convention (United Nations, 2010) of which South Africa is a contracting party. An ecosystem-based approach to MSP is founded on ecosystem health, whereas an integrated-use approach to MSP is underpinned by economic growth (Qiu and Jones, 2013).

Global economic powers are currently interested in promoting East Africa's oceans economy via programmes such as China's One Belt One Road Initiative and Japan and India's Asia Africa Growth Corridor (Buys, 2018). South Africa's current plans to grow its oceans economy have created high expectations amongst many stakeholders, and in response to this socio-political context, we have developed a research agenda to address key challenges in the country's emerging MSP process (**Table 1**). From our perspective, as marine scientists, the most urgent research challenges to address fall within three broad themes. First is a critical need for an improved understanding of the cumulative impacts of human activities (including climate change) on marine ecosystem structure and function. Existing activities are intensifying, and new activities are emerging (e.g., seabed mining) and we do not know how these will impact ecosystem health or the delivery of ecosystem services. Second is the need for interdisciplinary research to inform and support integrated ocean management, including management strategies that are dynamic in space and time and can measure trade-offs between competing human activities (see, for example, Harris and Lombard, 2018). Third is the requirement for the development of robust scenario-planning tools to aid multi-sector decision making and manage conflict. We believe these challenges are applicable in all socio-economic contexts, particularly in least-developed countries with strong and urgent economic growth imperatives. We recognize that many additional challenges exist and our aim is not to be comprehensive, but merely to share our specific approaches to these challenges. Our research agenda aims to support sustainable MSP practices that do not transcend environmental tipping points or safe operating spaces for human well-being (Rockström et al., 2009; Barfuss et al., 2018). Here we describe seven research projects that specifically address components of these three broad themes, and discuss the potential of our preliminary results to advance an ecosystem-based approach to MSP.

MEASURING ECOSYSTEM CONDITION

An ecosystem-based approach to MSP is predicated on the principle of achieving good ecological status for oceans (WFD, 2000). However, *in situ* measurements of ecosystem condition at broad scales remains a global knowledge gap as does a better understanding of the cumulative impact of human uses on ocean ecosystems (Borja, 2014; Ehler et al., 2019). Under Operation Phakisa, South Africa's national interest in growing its oceans economy is outpacing the development and application of robust tools to measure marine ecosystem condition. Most important is the absence of a standardized data-derived approach. This poses challenges for national assessments, for example, the National Biodiversity Assessments (NBAs) that are conducted approximately every 7 years[2] and contribute to the environmental monitoring requirements of national government. The NBA currently relies on a proxy of cumulative human impacts to infer good, fair or poor ecosystem condition at a national scale, creating the need for fine-scale ecosystem-based condition assessments using ecological data (Sink et al., 2012a).

We are presently engaged in a benthic research project designed to validate the current proxy method. Potential indicators of ecosystem condition have been identified using demersal fish and benthic invertebrate data from rocky reef ecosystems on South Africa's east coast. This project also aims to improve the understanding of the effects of cumulative anthropogenic pressures on the structure and function of marine ecosystems. Species composition and abundance and species biological traits are being assessed to determine the functional structure of the ecosystem. Traits were chosen to represent the trophic ecology and life history of the community (for example, fish biological traits included maximum depth, maximum length, trophic level, longevity, reproductive mode and guild, gregariousness, feeding guild, habitat preference, water column position, activity, and mobility).

This benthic study tests a suite of structural (species richness, Shannon Weiner diversity, total abundance, and biomass) and functional metrics (average and proportion of biological traits and community metrics of functional richness, functional evenness, functional diversity, functional specialization, functional originality, and Rao's quadratic entropy). Study

[1] www.operationphakisa.gov.za

[2] https://www.sanbi.org/

TABLE 1 | A summary of six research projects designed to address key challenges in advancing an ecosystem-based approach to marine spatial planning (MSP) under economic growth imperatives in South Africa.

Key challenge	Research focus	Approach	Tool	Output	Advancement	Impact
1. Measuring ecosystem condition	Rocky reefs; microbial communities	Structural and functional approach using biological and genetic traits of demersal fish, benthic invertebrates, and microbial communities	Remote imagery (stereo-baited underwater video systems); molecular taxonomy and molecular microbial community analysis	Indicators for measuring ecosystem condition; standardized method for national condition assessments	*In situ* measurement of ecosystem condition to test surrogates based on cumulative human use; incorporation of microbes in national biodiversity assessments; development of sensitive tools for assessing ecosystem health, anthropogenic impact and responses to climate change	National Biodiversity Assessment products prepared every 7 years for government; foundational biodiversity contribution to the marine spatial plan for Algoa Bay
2. Modeling impacts of climate change on food webs and fisheries	Southern Benguela ecosystem	End-to-end models	Atlantis and NEMO-MEDUSA 2.0 modeling frameworks	Ecological and fishery indicators of ecosystem-level responses to the impacts of climate change	Consideration of the likely impacts of climate change in fisheries and resource management	Proof of concept for the evaluation of the cumulative impacts of climate change on the structure and functioning of the southern Benguela ecosystem
3. Managing fisheries with an ecosystem approach	Spatial fisheries management	Review of legislation and existing spatial management measures	n/a	Identification of legal instruments that support spatial fisheries management	Integration of existing spatial management instruments into fisheries management to support MSP	Recommendations for implementing Priority Fishing Areas and Fisheries Management Areas under the Marine Living Resources Act
4. Using dynamic ocean management to resolve conflicts between fisheries and threatened species	Seabirds, small pelagic fish, and fishing industries	Experimental and multi-disciplinary approach (biological, social and economic sciences)	GPS tracking, acoustic surveys, questionnaires	Scientific evidence of the benefits of marine protected areas (MPAs) for marine top predators	Incorporation of dynamic management of a MPA into conflict resolution between top predators and fisheries	Proof of concept for the feasibility of dynamic ocean management for small pelagic fishing industries
5. Managing conflicting objectives in a growing marine tourism industry	Boat-based whale-watching (BBWW) industry	Multi-disciplinary approach (biological, social and economic sciences); systems analysis	Land- and boat-based observations; questionnaires; economic evaluation; system dynamics (SD) modeling	SD model of the BBWW industry in Plettenberg Bay	Scientifically determined species-specific permit regulations; identification of tipping and leverage points for sustainable industry management	Updated National legislation for BBWW industry; proof of concept for using SD models to promote sustainable practices for marine tourism
6. Developing scenarios for alternative management strategies in complex marine social-ecological systems	Algoa Bay social-ecological system	Systems analysis	SD modeling	MSP decision-support tool and scenarios for different marine management options for Algoa Bay created with SD models	Incorporation of systems analysis and SD modeling into MSP	Case study and proof of concept for the future development of SD models in marine area plans required by new legislation

sites are distributed along a cumulative pressure gradient, using the Pondoland no-take marine protected area (MPA) to provide a reference for good condition rocky reef ecosystems. Using fish data collected from stereo baited underwater video systems, we have been able to show that structural metrics are not sensitive to changes in cumulative pressures. In contrast, functional evenness, functional originality, mean and maximum size, mean trophic level, mean longevity and proportion of higher carnivores show a significant decline in response to increasing cumulative pressures. Multivariate analyses also identify which traits are most vulnerable to cumulative pressures, based on site differences between the no-take MPA and heavily exploited localities. These results emphasize the need to assess both the structural and functional response of biological communities to human pressures and have identified initial indicators that can be used to measure the condition of other rocky reef ecosystems nationally. The study thus provides the first validation of the proxy method used to generate national maps of marine ecosystem condition in the NBA.

In addition to broad-scale biodiversity assessments such as the NBA, there is a need for continuous monitoring of ecologically and economically important habitats that can provide early warning signs of threats to ecosystem functioning. Microorganisms account for up to 70% of the total biomass of the oceans (Bar-On et al., 2018), and play a critical role in driving global biogeochemical cycles (Azam et al., 1983). Community analysis of microbial biomass reveals extraordinary taxonomic diversity with distinct communities in different water masses (Sunagawa et al., 2015; Venkatachalam et al., 2019). Their diversity is reflected by metabolic versatility that allows marine microorganisms to respond rapidly to changes in their physical and chemical environment, including anthropogenically-driven change (Fuhrman et al., 2015). These responses, including shifts in the diversity and structure of microbial communities and their metabolic activity, can be used as a sensitive tool for assessing ecosystem health, anthropogenic impact and responses to climate change (Matcher et al., 2011).

We have employed this approach in a comparative study of the microbial communities of three estuaries, two of which are impacted by human activity, in Algoa Bay on the southeast coast of South Africa. The study showed conservation of the microbial communities of the marine-dominated compared with the freshwater-dominated estuaries and a drop in bacterial species richness in the human impacted systems. In addition, there was a significant difference in the response to the impact of agricultural versus urban activity in the river catchments and in the estuaries (Matcher et al., 2018). We are using this bottom-up approach to characterize the pelagic and benthic microbial communities of Algoa Bay, including the Sundays and Swartkops River estuaries. The data provide insight into macro- and mesoscale variability that reflects the complexity of freshwater and marine influences and anthropogenic impact with important implications for the development of a marine spatial plan for the Bay (see Dorrington et al., 2018). This local-area plan can inform the broader Marine Area Plans required by South Africa's emerging MSP legislation.

MODELING IMPACTS OF CLIMATE CHANGE ON FOOD WEBS AND FISHERIES

Climate change is a globally-recognized threat to marine fisheries (Cochrane et al., 2009), and of the seven broad categories of human use of the ocean, fisheries are considered to be the most vulnerable to a range of both primary and secondary drivers of climate change (Santos et al., 2016). Within South Africa, a trait-based assessment of the likely sensitivity of South African species to climate change identified that endemic and/or species with a threatened or depleted stock status ranked among the most sensitive to climate change (Ortega-Cisneros et al., 2018b). Potts et al. (2015) identified a knowledge gap regarding the predicted impacts of climate change on coastal fishes, and range shifts in economically significant species (Fairweather et al., 2006; Blamey and Branch, 2012) could be indicators of the impacts of climate change, but significant new knowledge will be required to determine if this is indeed the case. The South African National Department of Agriculture, Forestry and Fisheries (DAFF) is developing a draft Climate Change Mitigation and Adaptation Plan for marine fisheries and aquaculture. We have responded to this knowledge gap by developing models of the potential impacts of climate change on food webs.

Climate projections indicate that an increase in sea surface temperature (SST) in South African waters of approximately 3°C is expected by 2099 compared to SST observed in 2000 under the Representative Concentration Pathways (RCP) 8.5 emission scenario of the Intergovernmental Panel on Climate Change (Popova et al., 2016). Early signs of surface and shelf bottom water aragonite undersaturation are also expected by the end of the century in South Africa (Popova et al., 2016), however, this can be observed earlier in areas influenced by upwelling such as the west coast of South Africa (Gruber et al., 2012). Anthropogenic climate change (e.g., ocean warming) has been shown to have a wide range of effects on marine organisms (e.g., Edwards and Richardson, 2004; Hoegh-Guldberg et al., 2010) with consequences on catch potential (e.g., Hays et al., 2005; Cheung et al., 2011) and livelihoods (Barange et al., 2014; Pecl et al., 2017). Furthermore, sensitivity to acidification is intensified when taxa are simultaneously exposed to increased seawater temperature (Kroeker et al., 2013; Nagelkerken and Connell, 2015). The warming and acidification forecast for South Africa can be expected to have profound effects on a number of species and potentially on whole ecosystems.

Climate projections from the coupled "NEMO – MEDUSA 2.0" model (Yool et al., 2013) under the RCP 8.5 emission scenario and the "Atlantis on the Benguela and Agulhas Currents" model (ABACuS v2, Ortega-Cisneros et al., 2017) were used to simulate the effects of ocean acidification and warming in the southern Benguela system. Model simulations represent the period 1990 to 2050. The individual and combined effects of

ocean acidification and warming were compared to the control run (no fishing, warming or acidification pressures) to determine the effect of these stressors on groups' biomass in the southern Benguela. In the simulations, because of the limited number of studies evaluating the effects of acidification on invertebrate and fish species in South Africa, acidification is assumed to affect the mortality of phyto- and zooplankton groups only. Other invertebrate groups such as macrobenthos, meiobenthos and squid as well as fish groups were assumed not to be affected by acidification.

The combined effects of warming and acidification resulted in biomass reductions for most plankton and fish groups. Several target species such as anchovy *Engraulis encrasicolus*, deep-water hake *Merluccius paradoxus* and shallow-water hake *M. capensis*, snoek *Thyrsites atun* and sardine *Sardinops sagax* were also negatively affected by the combined effect of warming and acidification. The observed impacts of acidification on most model groups were explained by indirect effects since acidification affected only plankton groups in our simulations. However, the impacts of warming were mostly attributed to direct effects on the consumption, growth, mortality and reproduction of model groups (Ortega-Cisneros et al., 2018a).

This study has provided insight into the potential responses of marine species to climate change in South Africa. Our results suggest that several important target species are likely to be affected by climate change, which will potentially have repercussions on the country's economy. Our study also identified information gaps and research priorities, which is important in countries where resources and available information are limited. The consideration of the likely future impacts of ocean acidification and warming, and its combined effects with other anthropogenic stressors, is essential to better inform resource management and planning in South Africa, especially in the currently-emerging MSP processes that will need to remain adaptive to the spatial uncertainties resulting from climate change.

MANAGING FISHERIES WITH AN ECOSYSTEM APPROACH

In 1998, an ecosystem-based fisheries management (EBFM) approach was adopted in South Africa with the enactment of the Marine Living Resources Act 18 of 1998 (MLRA). Subsequently, some ecosystem-based measures were incorporated into the permit conditions of most commercial fisheries sectors, but progress with implementation has been slow. While an EBFM approach is implicitly considered in scientific and management working groups, the implementation of explicit measures remains to be demonstrated (Hutchings et al., 2009; Petersen et al., 2010). A further deficiency in the implementation of EBFM in South Africa is the incorporation of spatial management instruments. Some spatial and temporal measures are in place, however, the rationale for the existence of many of these measures has not been explicitly stipulated and there have been calls to review and improve such measures (Shannon et al., 2006; Sink et al., 2012b). There are also concerns that expanding and diversifying mining interests may compromise the food and job security provided by fisheries (Norman et al., 2018).

The sustainable management of natural resources requires an understanding of both the spatial and temporal scales over which the social-ecological system operates (Ostrom, 2009). Although many temporal measures exist to manage fisheries globally, the spatial nature of ecosystems, natural resources and human activities intuitively requires that management measures should incorporate spatial strategies (Crowder et al., 2008). Spatial aspects of fisheries that may benefit from area-based management measures include stock structure (Reiss et al., 2009) and catch distributions (Rassweiler et al., 2012), key areas for life history stages (Fisher and Frank, 2002) (e.g., areas for spawning, nursery, migration, life history cues), bycatch hotspots (Witherell and Pautzke, 1997), and user conflict (Kaiser et al., 2000). The EBFM approach is fundamentally a spatially-explicit approach, and ocean zoning and MSP form a crucial part of this concept (Pikitch et al., 2004).

In order for South African fisheries management to fully adopt an ecosystem-based approach, the full range of potential spatial management instruments needs to be identified and considered for implementation. We have investigated spatial management options by: (i) reviewing the existing legal environment that supports spatial ocean management in South Africa; (ii) identifying potential legal instruments that are or may be implemented to support spatial fisheries management; (iii) exploring the manner in which these instruments may be implemented; and (iv) providing research recommendations to improve integration of the spatial management instruments into fisheries management. Eight Acts and Bills were examined, including the National Environmental Management Act 107 of 1998 (NEMA), National Environmental Management: Integrated Coastal Management Act 24 of 2008 (NEM:ICMA), National Environmental Management: Protected Areas Act 57 of 2003 (NEM:PAA), National Environmental Management: Biodiversity Act 10 of 2004 (NEM:BA), the Mineral and Petroleum Resources Development Act 28 of 2002 (MPRDA), the Marine Living Resources Act 18 of 1998 (MLRA), the Aquaculture Development Bill B 22-2018 and the Marine Spatial Planning Bill B 9D-2017.

From the Acts and Bills examined, 11 spatial legal instruments have been identified that can be used to improve area-based management in the ocean, including measures that could be used to implement MSP. Results show that nine instruments support area-based management of South Africa's fisheries, however, only two have been implemented. Collectively, these instruments represent opportunities to improve spatial fisheries management, for example, spatial measures currently implemented through permit conditions may be formalized through the implementation of legal instruments. Some existing spatial closures could be considered or strengthened as Other Effective Area Based Conservation Measures (OECMs) recognized by the Convention on Biological Diversity. With the elevation of the MSP Bill to the Presidency for enactment, there is an incentive to formalize spatial management

measures to facilitate the MSP process. In particular, it is recommended that the identification and implementation of Fisheries Management Areas and Priority Fishing Areas under the MLRA be prioritized to support an EBFM approach to fisheries, and the development of an ecosystem-based approach to MSP.

USING DYNAMIC OCEAN MANAGEMENT TO RESOLVE CONFLICTS BETWEEN FISHERIES AND ENDANGERED PENGUINS

No-take zones can be important tools within an ecosystem-based approach to achieve sustainable fishing and re-establish ecosystem integrity (Roberts et al., 2005). However, their potential benefits for vagile species such as small pelagic fish and top predators remain questionable. In addition, they are generally designed with definite sizes and shapes, although these can be dynamically managed in space or time. For example, high-risk areas for bycatch of yellowtail flounders *Limanda ferruginea* by a New England scallop fishery are identified and updated daily, which allows the lifting of the closed seasons previously in place, thereby adding $10 million to the fishery each year (O'Keefe and DeCelles, 2013). Maxwell et al. (2015) provide additional examples of successful dynamic ocean management measures.

In South Africa, the population of the endemic African penguin *Spheniscus demersus* has halved since 2004 (Crawford et al., 2011). They predominantly feed on sardines and anchovies which are also the target species of the purse-seine commercial fishery. Since 2008, a 20 km radius experimental purse-seine fishing exclusion has been initiated around two pairs of penguin colonies (Dassen and Robben islands on the West Coast, and St Croix and Bird islands in Algoa Bay on the East Coast, the latter supporting half of the global population) with alternating closure regimes in 3 years cycles (e.g., Pichegru et al., 2012; Sherley et al., 2018). Between 2012 and 2017, in addition to collecting information on the foraging performance and reproductive success of penguins in Algoa Bay, we conducted small-scale acoustic surveys to determine the relative abundance of pelagic fish around their colonies (see McInnes et al., 2015). We related these to the fishing exclusion patterns, while controlling for monthly environmental conditions. We also compared fishing patterns (costs in terms of travel time and benefits in terms of size of landings) during and outside closures to estimate the potential socio-economic impact of no-take zones to the industry.

The results show that the costs of the closure remained low to the fishing industry and catches remained stable through time, despite variability in fish abundance. By contrast, fishing exclusions largely benefitted penguins in terms of breeding success and foraging performance, particularly during periods of poor environmental conditions. Consequently, interviews were conducted with stakeholders in the fishing industry, initiating a discussion platform between fishers and scientists, to explore if a dynamically-managed, adjustable no-take zone in Algoa Bay can better protect penguin food sources, while still considering fisher economies (Ginsburg et al., 2018). Monthly acoustic surveys assessing fish abundance could be used to suggest size and timing of fishing exclusions, thereby allowing fishers additional operational areas when fish stocks are high. Our research provides a proof of concept for spatially and temporally dynamic management practices, as well as the co-development of these practices with resource-dependent stakeholders. Given that the current MSP process in South Africa lacks a platform for meaningful engagement with civil society (Reed and Lombard, 2017), our research outputs provide an encouraging example of the potential for win-win outcomes that can be achieved with purposeful stakeholder consultation.

MANAGING CONFLICTING OBJECTIVES IN A GROWING MARINE TOURISM INDUSTRY

Coastal and marine tourism have the potential to make an important contribution to the blue economy. Boat-based whale-watching (BBWW) has become one of the fastest growing marine tourism industries worldwide, benefitting many communities and countries, and affecting many nearshore cetacean populations (O'Connor et al., 2009). BBWW was initially viewed as a sustainable and non-consumptive use of cetaceans (Allen, 2014), and a profitable replacement for commercial whaling (Hoyt and Hvenegaard, 2002). More recent research has shown the potential for BBWW to have extensive negative impacts, including local extinctions (Lusseau et al., 2006; Parsons, 2012; Senigaglia et al., 2016). In the case of tourism practices that rely on human-wildlife encounters, long-term sustainability requires the continued presence of wildlife, and ethical codes of conduct require empirical evidence to set specific standards.

The Indian Ocean Rim Association (IORA) recognizes the vital contribution of marine tourism to the blue economy (see Rogerson et al., 2018). South Africa, currently serving as the IORA chair, is at the forefront of promoting the inclusion of marginalized groups in marine tourism, with a focus on BBWW. The South African BBWW industry was formalized in 1998, requiring permits to operate under a framework of regulations and a voluntary code of conduct. The Department of Environmental Affairs (DEA) is responsible for administering the permits and enforcing the regulations. Globally, the South African BBWW industry is perceived as one of the most sustainable of its kind, owing to the strong scientific basis from which the regulations and code of conduct were designed. However, an assessment of the efficacy of these regulations to protect all targeted species has not been done and species-specific regulations may be needed for the more vulnerable and less conspicuous populations e.g., the Endangered humpback dolphin (*Sousa plumbea*) and Vulnerable inshore Bryde's whale (*Balaenoptera edeni*) (Penry et al., 2016; Plön et al., 2016).

Additionally, increasing pressure to expand the industry for economic growth and social upliftment under Operation Phakisa, requires a thorough understanding of the resource value, it's socio-economic potential, and potential impacts on target species.

In 2005, an economic assessment by Turpie et al. (2005) established that the BBWW industry contributed approximately R37 million to South Africa's Gross Domestic Product annually. Their report supported the growth of the industry based on the potential economic benefits, suggesting that the number of operators could be increased by up to 40% "in the near future." Since then studies on the growth and subsequent re-evaluation of the economic importance of this industry to South Africa have been lacking. Although the socio-economic benefits of whale and dolphin tourism and growth of the industry are apparent, with an increase from 20 to 40 available permits between 1999 and 2017 (Department of Environmental Affairs [DEA], 2017a,b), the benefits must be considered in light of the dependence of the industry on the availability of a resource in the form of regular and reliable sightings of whales and dolphins. This increase was based on the recognition of the potential economic and social benefits of the industry, but with little or no consideration of the ecological factors on which the industry is dependent. Sustainable BBWW depends on continuous monitoring of the impacts on cetaceans, compliance with enforceable regulations, commercial profitability, and tourist satisfaction (Hoyt, 2003; Corkeron, 2006).

A multi-disciplinary team is conducting research in Plettenberg Bay on the South coast of South Africa, to assess the current status of, and develop sustainable practices for, boat-based marine tourism. The strategic intent of the program is to support Operation Phakisa's initiative to promote the development of a sustainable oceans economy. The objectives are to: measure the behavioral responses of whales and dolphins to vessel approaches; assess the effectiveness of existing guidelines to mitigate potential negative impacts; determine rates of permit regulation transgression; quantify the direct and indirect economic value of the industry; and determine levels of customer satisfaction in relation to their perceptions and expectations based on marketing of the industry. The overall goal is to develop a sustainable marine tourism sector with minimal impact on the resource that it depends upon. We are using a systems analysis approach with system dynamics (SD) modeling tools to identify tipping points at which the industry becomes economically or ecologically unsustainable, and leverage points where management or policy interventions could move the system onto a sustainable path. SD models allow diverse stakeholders to contribute both qualitative and quantitative information, and to see the impact of different decisions on their variables of interest (e.g., annual earnings or whale population numbers). The Plettenberg Bay project is providing a platform for future group model-building exercises in a multi-stakeholder environment, drawing on biophysical and socio-economic research methods. From our perspective, multi-disciplinary research projects of this nature have the highest likelihood of success in the management of conflicting objectives in a growing oceans economy.

DEVELOPING SCENARIOS FOR ALTERNATIVE MANAGEMENT STRATEGIES IN COMPLEX MARINE SOCIAL-ECOLOGICAL SYSTEMS

Marine environments and marine governance structures are both complex systems, and management strategies often fail to encompass this complexity (Hazen et al., 2018; Lewison et al., 2018; Lombard et al., in press). Integrated management approaches should therefore supplement sector-specific approaches and should adopt methods of analysis and decision-support tools that recognize multiple forms of complexity (Douvere, 2008; Lombard et al., 2019). There is a growing interest in complexity research globally, given the increasing impacts of humans on the biosphere (Steffen et al., 2015), and MSP processes can benefit from a paradigm shift from linear management approaches, to more adaptive approaches that consider uncertainties, feedbacks, and plausible future scenarios. Systems analysis, an approach that addresses complexity, can provide a framework to understand key dynamic interactions, feedbacks and unintended consequences across multiple sectors (Pongsiri et al., 2017) and is therefore well suited to MSP (Wang et al., 2014; Boumans et al., 2015). SD modeling, a structured approach to systems analysis, is a rigorous method for modeling complex systems and building computer simulations which can assist with understanding the behavior of systems under different conditions and future scenarios (Sterman, 2000; Ford, 2009; Deenapanray and Bassi, 2015). SD models can integrate environmental, social and economic components within the marine system and therefore provide holistic decision-support to understand the varying levels of complexity (Deenapanray and Bassi, 2015).

Weller et al. (2014, 2016) used SD models to better understand the impacts of multiple drivers on endangered African penguin populations on two islands in South Africa. Their model scenarios identified which management strategies were likely to provide the most benefit for penguin populations. We are using a broadly similar modeling approach to develop scenarios for multiple sectors in Algoa Bay, South Africa, and how these sectors will (1) be impacted by external drivers (e.g., harmful algal blooms, range shifts in commercial species) and (2) impact one another as they respond to external drivers (e.g., an increase in fishing effort for small pelagic species may impact top predators reliant on the same species). SD models will be developed with stakeholders to allow them to identify which mitigation and adaptation strategies will most likely reduce their sector's vulnerability to the negative impacts of external drivers, while also minimizing any negative interactions among sectors.

Our particular study forms part of a broader programme to develop a marine spatial plan for Algoa Bay (Dorrington et al., 2018). Given that the focus of MSP is inherently spatial, and that SD models provide temporal outputs (in the form

of behavior-over-time graphs), we are developing methods to "soft-couple" spatial and temporal outputs in a single decision-making framework. Soft coupling provides a trade-off between complexity and tractability, and our intention is to provide a proof of concept for SD modeling as a tool for MSP in least developed countries (that do not have the resources for fully integrated model coupling).

From our perspective, systems-based approaches can support more efficient planning and implementation processes and help to optimize policy interventions by identifying trade-offs and consequences related to different management strategies (White et al., 2012; Lester et al., 2013; Pongsiri et al., 2017). Adopting a complex systems analysis approach to MSP in South Africa could therefore provide a realistic research lens and could assist in informed and coordinated decision-making about the future management of the country's marine environment, aiming to ensure the rational exploitation of its marine resources and facilitate sustainable development of the marine economy.

CONCLUSION

Through its National Development Plan and Operation Phakisa, South Africa has a strong commitment to grow its oceans economy. At the same time, the country's Marine Spatial Planning Bill is based on the principles of ecosystem-based MSP, as opposed to integrated-use MSP. Careful navigation will be required to ensure that short-term gains in one sector will not outweigh concerns about long-term sustainability in other sectors, and that all sectors will have equal power at the negotiating table. As marine scientists, we need to identify what sort of scientific information will be most useful, and have most traction, at the policy level. To this end, we have adopted a multi-disciplinary approach to address what we believe are the key challenges in advancing an ecosystem-based approach to MSP in a political climate of economic growth imperatives.

The outputs of projects described in this manuscript are designed to assist at the ecosystem assessment and monitoring level (with data-derived methods to measure ecosystem condition), the scenario planning and stakeholder engagement level (with climate, ecosystem, and SD models), the management level (particularly of resources that require dynamic management), and the policy level (by developing evidence-based codes of conduct for tourism, and informing and applying existing legal instruments for spatial fisheries management). Ultimately, the hope is to advance a more integrated and adaptive approach to ocean management in South Africa, using tools that can be applied in countries with similar socio-political and environmental contexts. Recognizing that our research projects address only some of the many challenges in ecosystem-based approaches to MSP, we are continuing to develop new programmes to address additional challenges (e.g., the impact of seabed mining on fisheries, and the impact of ocean noise on ecosystem and species). Nonetheless, we hope that we can add to the global research knowledge base, and other MSP processes, with the lessons we have learned thus far.

DATA AVAILABILITY

The datasets for this study will not be made publicly available because the data are part of students theses. The theses will be published online in due course.

AUTHOR CONTRIBUTIONS

AL conceptualized idea for research, wrote manuscript with case study inputs from co-authors, post-graduate supervisor of five of the co-authors, post-doctoral host of two co-authors. RD, KO-C, GP, LP, JR, KPS, EV, MW, and KJS wrote a subsection of the manuscript. TG and AM conducted analyses for a subsection of the manuscript. All authors provided final editorial input.

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Practical Approaches and Advances in Spatial Tools to Achieve Multi-Objective Marine Spatial Planning

Amanda T. Lombard[1], Natalie C. Ban[2], Joanna L. Smith[3], Sarah E. Lester[4], Kerry J. Sink[1,5], Spencer A. Wood[6], Aerin L. Jacob[2†], Zacharoula Kyriazi[7], Richard Tingey[8] and Helena E. Sims[9]*

[1] Institute for Coastal and Marine Research, Nelson Mandela University, Port Elizabeth, South Africa, [2] School of Environmental Studies, University of Victoria, Victoria, BC, Canada, [3] Nature United, The Nature Conservancy, Fredericton, NB, Canada, [4] Department of Geography, Florida State University, Tallahassee, FL, United States, [5] South African National Biodiversity Institute, Pretoria, South Africa, [6] School of Environmental and Forest Sciences, University of Washington, Seattle, WA, United States, [7] Marine Biology Research Group, Faculty of Sciences, and Maritime Institute, Department of Public International Law, Ghent University, Ghent, Belgium, [8] Spatial Support Systems LLC, Salt Lake City, UT, United States, [9] The Nature Conservancy, Victoria, Seychelles

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***Correspondence:**
Amanda T. Lombard
mandy.lombard@mandela.ac.za

†Present address:
Aerin L. Jacob, Yellowstone to Yukon Conservation Initiative, Canmore, AB, Canada

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Marine spatial planning (MSP) processes seek to better manage ocean spaces by balancing ecological, social and economic objectives using public and participatory processes. To meet this challenge, MSP approaches and tools have evolved globally, from local to national scales. At two International Marine Conservation Congresses (2016 and 2018), MSP practitioners and researchers from diverse geographic, technical and socio-economic contexts met to share advances in practical approaches and spatial tools to achieve multi-objective MSP. Here we share the lessons learned and commonalities that emerged from studies conducted in Belize, Canada, South Africa, Seychelles, the United Kingdom and the United States on a number of topics related to advancing MSP. We identify seven important themes that we believe are broadly relevant to any multi-objective MSP process: (1) indigenous and local knowledge should inform planning goals and objectives; (2) transparent and evidence-based approaches can reduce user conflict; (3) simple ecosystem service models and scenarios can facilitate multi-objective planning; (4) trade-off analyses can help balance diverse objectives; (5) ecosystem services may assist planning for high value-data poor Blue Economy sectors; (6) game theoretic decision rules can help to deliver fair, equitable and win–win spatial allocation solutions; and (7) strategic mapping products can facilitate decision making amongst stakeholders from different sectors. Some of these themes are evident in MSP processes that have been completed in the previous decade, but the fast-evolving field of MSP is addressing increasingly more complex objectives, and practitioners need to respond with practical approaches and spatial tools that can address this complexity.

Keywords: ocean zoning, marine conservation, trade-offs, decision support, scenario planning, ecosystem services, integrated ocean management

INTRODUCTION

Over the last decade, marine spatial planning (MSP) has become an increasingly accepted approach to achieve multiple objectives for ocean management. At least 13 countries have approved marine plans covering 7% of the world's Exclusive Economic Zones and Territorial Seas. By 2025, marine plans may be implemented in more than 40 countries around the world including several Small Island Developing States (SIDS) (Smith, 2017). MSP presents several significant challenges including choosing appropriate data, models and decision support tools to inform the planning process. Advances in approaches to, and spatial tools for, multi-objective marine planning are necessary to address particular challenges posed by the different spatial, temporal and socio-economic scales of uses and activities in a given planning context (De Santo, 2013). To date, almost 100 decision-support tools for MSP have been developed (Beck et al., 2009; Bolman et al., 2018) but there is varied success for using these tools during real-world planning processes, particularly in data-poor geographies and SIDS (Pınarbaşı et al., 2017).

Given that MSP is a public and participatory process to address ecological, social, and economic objectives with stakeholders in a transparent way, decision-support tools need to be able to estimate, visualize, and evaluate trade-offs among overlapping uses or conflicts among activities. The science and practice of developing and using technical and spatial tools for MSP is evolving, including more explicit consideration of ecosystem services (Arkema et al., 2015) and approaches to conflict analysis and management. Fast-tracking of MSP processes globally has also provided an opportunity for spatial tools to advance in their capacity to address multiple objectives and move from sector-specific to multi-objective planning.

Here, we present recent advances in practical approaches and spatial tools from several ongoing marine planning efforts. These studies were presented during two special sessions at the Society for Conservation Biology's International Marine Conservation Congresses (IMCC) in 2016 and 2018 and involve a range of interdisciplinary approaches, contexts, and geographies. We provide case study examples from six countries, spanning seven broad themes relevant to any multi-objective MSP process. In Canada (British Columbia), we demonstrate that indigenous and local knowledge should inform planning goals and objectives, and that evidence-based approaches can reduce user conflict; in Belize, we show how simple ecosystem service models and scenarios can facilitate multi-objective planning; in the United States (California), trade-off analyses have helped balance diverse objectives; in the Seychelles, we demonstrate how ecosystem services may assist planning for high value-data poor Blue Economy sectors; in the United Kingdom, we used game theoretic decision rules to help deliver fair, equitable and win–win spatial allocation solutions; and in South Africa, we show how strategic mapping products can facilitate decision making amongst stakeholders from different sectors.

Although the approaches presented here are not comprehensive and do not represent a systematic review of all active processes around the world, they reflect an assortment of actual on-the-ground experiences that we believe are broadly relevant and can contribute to the evolution of MSP today.

INDIGENOUS AND LOCAL KNOWLEDGE SHOULD INFORM PLANNING GOALS AND OBJECTIVES

People who live and work on or near the ocean observe changes over their lifetimes and can also accumulate generations of knowledge regarding previous baselines of marine resources. In the case of Indigenous peoples, wisdom and practices are passed down through generations, for example, in the form of dances, stories, traditions, and Indigenous laws (Berkes, 2018). This local and Indigenous knowledge should be considered paramount to informing MSP goals and objectives. Our research illustrates this point.

We developed community–academic research partnerships (Ban et al., 2018) to identify changes in size and abundance over the past 50 years of some focal species in order to inform MSP and fisheries management. The partnerships were created at the request of four First Nations (Indigenous peoples) on the central coast of British Columbia, Canada. Two species were of particular concern to them (Dungeness crab, *Cancer magister*; and Yelloweye rockfish, *Sebastes ruberrimus*) because these species are culturally important yet also targeted by commercial and recreational fisheries. Although our research focused on these two species, the methods are applicable to any species. We used a mixed methods approach, combining semi-structured interviews with ecological surveys and modeling to gauge the changes in these species between peoples' living memories (i.e., the first-time participants remember fishing for these species in their youth or early adulthood) and recent years. Ecological surveys and stock assessments either did not exist at all or were started only in the 2000s. We found that size (Yelloweye rockfish) and abundance (both species) had declined substantially, and that in some cases First Nations were unable to meet their needs for their constitutionally protected right to fish for food and for social and ceremonial purposes (Ban et al., 2017; Eckert et al., 2018). These results were brought by the First Nations partners to policy discussions, and they have resulted in changes to spatial management. More specifically, important crab fishing areas were closed to commercial and recreational fishing, and findings about the changes in sizes of Yelloweye rockfish are being incorporated into the latest assessment of this species of special concern.

These studies illustrate the importance of local and Indigenous knowledge in informing goals and objectives in marine planning. As is commonly the case globally, scientific surveys of these and other important species either did not exist or were started only recently. Without the information gathered from interviews, shifted baseline (e.g., significantly reduced biomasses of important species) might have been used to set default objectives (Pauly, 1995) in the absence of historical information. The community–academic partnerships were an effective *trans*-disciplinary approach to filling the data gap

and engaging people in thinking about future scenarios for these species.

TRANSPARENT AND EVIDENCE-BASED APPROACHES CAN REDUCE USER CONFLICT

On Canada's North Pacific Coast, the Province of British Columbia (BC) and 17 Coastal First Nations recently co-developed marine spatial plans to support sustainable economic development and a healthy marine environment across more than 100,000 km^2.[1] There were four sub-regional planning areas: Haida Gwaii, the North Coast, the Central Coast, and North Vancouver Island, together comprising the North Pacific Coast of BC. But effective implementation of each marine plan requires evaluating how key marine uses interact, including linkages on land and under global environmental change (Marine Plan Partnership for the North Pacific Coast [MAPP], 2016). We worked with First Nations on the Central Coast to identify priority income-generating activities in the Great Bear Sea and adjacent Great Bear Rainforest, identifying and modeling relevant abiotic and biotic conditions, to be used in an ecosystem services approach to evaluate potential environmental and economic synergies and trade-offs. The two highest-priority activities to emerge were shellfish aquaculture (for geoduck, scallops, and oysters) followed by nature-based tourism (bear-viewing); we focus here on shellfish aquaculture.

Shellfish aquaculture has the potential to meet local and regional objectives regarding income generation and employment while also supporting the global demand for seafood. While parts of southern BC have successful shellfish aquaculture industries, the Central Coast does not. However, shellfish has been important for food, social and ceremonial harvest for Coastal First Nations across the North Pacific Coast for millennia. Exploring the development of this industry (specifically Japanese scallop, *Patinopecten yessoensis*; and geoduck clams, *Panopea abrupta*) was identified as a top priority among Coastal First Nations, provided it did not negatively affect other activities including established, growing and potential industries such as forestry or nature-based tourism, and that it would remain viable with changing ocean conditions (reviewed in Holden et al., 2019).

We defined the range of abiotic conditions for successful scallop and geoduck aquaculture, to help identify the suitable natural locations to optimize growth and minimize mass mortality events. Through interviews with members of the scallop and geoduck aquaculture industry and researchers, we identified the tolerable and most favorable parameters for substrate, depth, temperature, salinity, tidal speed (both species), productivity (geoduck) and wave height (scallop) (Lancaster, 2017). This information was used to inform habitat suitability models for both species in the Central Coast and would be used to compare current aquaculture zoning to areas and variables important for nature-based tourism (e.g., visual quality, beach access and locations to see bears).

We are using our results to recommend zoning that minimizes potential conflicts and maximizes compatibilities in linked marine, coastal and terrestrial environments. Combined with community engagement, this iterative process can adaptively manage multiple uses and activities to support human well-being, governance and ecological integrity.

SIMPLE ECOSYSTEM SERVICE MODELS AND SCENARIOS CAN FACILITATE MULTI-OBJECTIVE PLANNING

Marine spatial planning processes are demonstrating how scientifically credible models and maps of ecosystem service production are helpful for balancing competing uses such as tourism, renewable energy, and commercial fisheries (Guerry et al., 2012; Arkema et al., 2015; Ruckelshaus et al., 2015). Ecosystem service modeling typically begins by quantifying the risks of human activities to the structure and function of natural habitats (Arkema et al., 2014), followed by modeling the benefits provided by natural habitats for people's livelihoods and well-being through ecological production functions. By pairing multiple ecosystem services and metrics with spatially explicit scenarios that compare alternative management options, it is possible to highlight how proposed marine spatial plans create synergies and trade-offs among activities in space.

This approach to ecosystem service modeling was exemplified during the creation of the Belize Integrated Coastal Management Plan (Coastal Zone Management Authority and Institute [CZMAI], 2016). During the planning, teams of researchers, practitioners, and stakeholders evaluated how human impacts on coral, mangrove, and seagrass habitats would change the potential for ecosystems to provide coastal protection, tourism and lobster fishery benefits under three alternative management scenarios that promoted either habitat conservation, coastal development or "informed management" (Arkema et al., 2015). As a result, the plan explicitly considers how coastal management can provide benefits to multiple sectors and stakeholders, given their local visions and values (Verutes et al., 2017).

One important lesson to emerge from recent planning efforts is that simple process-based ecosystem service models and spatial tools can be more useful than traditional heuristic models. Simple quantitative models help planners prioritize information-gathering, build local capacity and align stakeholders and appropriate authorities (Rosenthal et al., 2015; Verutes et al., 2017). Furthermore, simple models allow for an iterative science and policy process—in which scientists, stakeholders, and decision-makers repeatedly re-evaluate proposed scenarios, predicted outcomes, and model assumptions throughout the planning process—and that can result in more robust marine plans (McKenzie et al., 2014). This iterative approach to co-creating scenarios,

[1] http://mappocean.org

science and knowledge can produce more credible, transparent and effective tools that resonate with governments and stakeholders.

TRADE-OFF ANALYSES CAN BALANCE DIVERSE OBJECTIVES FOR THE USE OF OCEAN SPACE

A marine spatial plan that uses predictive models and trade-off analysis can better balance diverse objectives for the use of ocean space, including development of emerging uses like offshore aquaculture and wind energy, while minimizing negative environmental impacts (Lester et al., 2013). Leveraging analytical models enables consideration of a broader array of concerns and goals, more objective decision-making and transparency around costs and benefits of different spatial planning options.

This assertion is demonstrated in a spatial planning analysis supporting the potential development of multiple types of offshore aquaculture in southern California (Lester et al., 2018b). The study developed spatial models of the predicted productivity and profitability of three marine aquaculture sectors (offshore kelp farms, offshore mussel farms, and finfish netpen farms), and linked these to spatial models of four existing sectors that represent important stakeholder concerns regarding aquaculture development: wild-capture fishery profits; the environmental health of the benthos given potential nutrient pollution from finfish farms; risk of disease outbreak among farms; and viewshed impacts from adding structures to the marine vista. These linked models were integrated with an analytical trade-off analysis that identified optimal spatial plans given a range of preference weightings for the different sectors. The analysis suggested thousands of optimal plans (depending on the preference profile), allowing value of individual sectors to be enhanced and negative impacts to be reduced relative to more conventional approaches to planning. For example, the analysis found that dramatic trade-offs are unavoidable only at very extreme levels of aquaculture development, and there are spatial planning options that would result in a significant new supply of seafood, providing billions of dollars in revenue, with small to no impact on existing sectors and the environment.

Although California has implemented a statewide network of marine protected areas (MPAs) through a process that included the use of predictive models and trade-off analyses to help balance conservation goals with fisheries objectives (Rassweiler et al., 2014), the region has not engaged in spatial planning for offshore aquaculture or other emerging ocean industries. However, this analysis has informed some discussions about potential future aquaculture developments in California, and if the region were to adopt a proactive spatial planning process that leveraged the analytical tools presented here, it could help to reduce the hurdles to development caused by regulatory uncertainty (Lester et al., 2018a). MSP can not only catalyze the development of a new industry, it can also safeguard that development follows a sustainable, rather than environmentally precarious, trajectory (Gentry et al., 2017). More generally, with an inevitable industrialization of the oceans on the near horizon around the world, multi-objective planning using predictive modeling and trade-off analyses can help achieve best-case outcomes.

ECOSYSTEM SERVICES MAY ASSIST PLANNING FOR HIGH VALUE-DATA POOR BLUE ECONOMY SECTORS

There is an important need in multi-stakeholder MSP to identify current conditions and trends by compiling information for each sector (Ehler and Douvere, 2009). By compiling the best available information and representing stakeholder preferences in a Geographic Information System, it is also possible to identify missing data. Data gaps usually exist because the questions that need answering for an integrated and ecosystem-based marine spatial plan have not been previously asked or answered. MSP processes are relatively rapid; thus it is challenging to fill these gaps during planning, but they must be addressed to ensure equity amongst stakeholders (Fox et al., 2013) and to assess trade-offs (Yates et al., 2015).

Some sectors, such as fisheries, use a common property resource and have a long history of regulation that requires collecting and sharing detailed information about their activities with managers (e.g., effort and catch value). These data often follow rigorous protocols that ensure confidentiality and can then be used to describe the sector's value within an economy. As a result, maps of activities, values and preferences are often available for decision-support tools to use to inform zoning, minimize impacts, and maximize benefits (Kenchington and Day, 2011; Agostini et al., 2015). In contrast, other sectors, such as tourism, rely on public and private resources and have very different reporting requirements that limit the type of data they are required to share about their activities. Obtaining access to these data for a marine spatial plan can be difficult because there are no or limited existing protocols to enable sharing and ensure confidentiality, which in turn creates challenges to represent high-priority areas for an equitable and transparent process.

In Seychelles, The Nature Conservancy is facilitating a MSP process on behalf of the government to expand marine protection, address climate change and support the Blue Economy for a 1.35 million km^2 area (Smith et al., 2018a).[2] More than 15 years of data were shared by the fisheries authority to create area-based values for that sector (Smith et al., 2018b). However, insufficient data were available for high-value tourism, the leading contributor to Seychelles' gross domestic product (World Travel and Tourism Council [WTTC], 2017).

To map high-value tourism, we measured the relative distribution and abundance of visitation throughout the 115-island archipelago based on the number of geotagged photographs shared on the Flickr social media website from 2005 to 2014 (Wood et al., 2013; Keeler et al., 2015). Using a recreation ecosystem service model, we observed that tourism was highest around the accessible beaches and dive sites on the main island. However, without additional data on how relative differences in visitation reflect absolute differences in user days or

[2]https://seymsp.com

expenditures, we lacked critical information for valuing tourism. The lack of long-term datasets and previous characterizations of the tourism sector also led to unanswered questions about the appropriate methods for defining ownership, administration and natural geographic boundaries. Nevertheless, this was an important first step in the process to represent high-priority areas for a significant sector in Seychelles' Blue Economy. Since these maps were produced early in the planning process, the MSP initiative provided opportunities to work with the marine tourism sector to fill data gaps. This, combined with our other experiences in Seychelles, leads us to conclude that spatial tools play many direct and indirect roles in the development of marine spatial plans by highlighting data gaps and supporting efforts to create a transparent, equitable decision-making process for all stakeholders.

BEYOND EFFICIENCY: GAME THEORETIC DECISION RULES CAN HELP TO DELIVER FAIR, EQUITABLE AND WIN–WIN SPATIAL ALLOCATION SOLUTIONS

Marine spatial planning processes may encounter circumstances where the coexistence between infrastructure projects and MPAs is a defined goal, but the goal may be unrealistic because the negative externalities from the proposed infrastructure on the MPA may be unavoidable. In such cases, the conflict that emerges cannot be resolved through mitigation but can potentially be resolved through some form of compensation (Elliott and Cutts, 2004). This issue has been examined using a hypothetical case study referring to a 50 km^2 MPA where there is an application of a 104 MW marine renewable energy (MRE) project. The input values were analogically adjusted from values found in reports and scientific articles about the United Kingdom part of Dogger Bank (Department for Environment Food and Rural Affairs [DEFRA] and Joint Nature Conservation Committee, 2011; Börger et al., 2014; GENECON, 2014). In this hypothetical case study, it was demonstrated that conflict arising owing to environmental externalities from MRE to the MPA can be resolved through compensation that must be agreed upon by at least two parties (e.g., the project developer and the MPA manager) through negotiations. However, in order for the negotiated compensation to be acceptable by both parties, two constraints should be met: (a) the lost utility from ecosystem loss in the MPA must be less than the gained utility from the infrastructure project, (b) the surplus from the agreement leaves both players better off than before the agreement (i.e., a win–win situation). Therefore, not only efficiency, but also fairness and equity can be achieved (Kyriazi et al., 2015). Efficiency is a state of resource allocation in which no individual or player can be better off without making at least one other player worse off. "How much better off" a player will be after the agreement depends on the size of that player's disagreement (walk away) point and whether he/she holds private information about it or not (Kyriazi et al., 2015). For instance, a "No Net Loss" compensation may be less than the MPA manager's disagreement point and hence an insufficient incentive for him/her to cooperate and reach an agreement (Lejano and Davos, 1999; Forest and The Katoomba Group, 2010). In this case, a "Net Gain" compensation may be preferable (especially if the manager is concerned about uncertain future impacts of the development on the MPA, or in cases where the goal for the MPA is ecosystem enhancement rather than maintenance). To overcome this, participation constraints should be set where both players should reveal their disagreement (walk away) points. Then, unique win–win solutions can be estimated using formal quantitative approaches such as co-operative game theoretic decision rules (Kyriazi et al., 2015, 2016) that fairly distribute the surplus from coexistence/cooperation among players, thereby resolving conflict. This approach has the following advantages:

- It prevents negotiation breakdown by avoiding asymmetric information exploitation and ensures transparency;
- Not only does it address externalities, but it also ensures a benefit (over the disagreement point), leaving both players better off;
- It limits the net gain of a player (e.g., the MPA manager) to a maximum, restricting him/her from potentially pursuing an unrealistic gain from the negotiated coexistence;
- It estimates a fair net gain in monetary terms, thereby overcoming the limitations of achieving a net gain through other already established methods (e.g., Flores and Thacher, 2002; Dunford et al., 2004; Zafonte and Hampton, 2007; Fischer et al., 2008). In this case, a goal of restoration and/or enhancement can be achieved by the MPA manager instead of the developer through the appropriate investment of the monetary compensation, thus making the whole compensation process more sustainable. Ultimately, the MPA's "win" can be translated as a conservation benefit, enhancing the positive reputation of the developer (for example through green branding) and demonstrating a win–win approach.

STRATEGIC MAPPING PRODUCTS CAN FACILITATE DECISION-MAKING AMONGST MULTI-SECTOR STAKEHOLDERS

Building on a long history of terrestrial conservation planning, South Africa has been undertaking marine biodiversity mapping, spatial assessment and systematic conservation planning since 2004 (see for example, Lombard et al., 2007, 2019; Harris et al., 2019). Here, we share experience from two National Biodiversity Assessments[3]; a 12-year process to develop a representative MPA network (Sink et al., 2012; Sink, 2016); and 4 years of work to support new national MSP legislation. We share the maps that were most useful and had the largest uptake from the hundreds of input data layers and analyses produced through this

[3]https://www.sanbi.org/

work to assess biodiversity, plan for protection and support MSP over the last 15 years. We also distil key elements in working and communicating with maps to support MSP and conservation uptake.

Essential maps that supported this work included a National Map of Marine Ecosystem Types; maps of spawning and nursery areas of commercial fisheries; maps of the distribution and intensity of human activity (to inform cost layers and provide spatial surrogates for ecosystem condition); sector-specific maps reflecting key fisheries challenges (such as bycatch and incidental mortalities); a map of existing spatial management measures (including MPAs); and maps reflecting ecosystem threat status and protection levels (SANBI and UNEP-WCMC, 2016; Kirkman et al., 2019). We found that a continually adapting process was a key requirement and allowed us to update maps to reflect increasing knowledge and changing biodiversity and industry priorities. Such flexibility is critical to allay fears of reluctant stakeholders in sharing their sector-specific spatial priorities (such as the mining sector sharing their current priorities that may change with increasing exploration and prospecting) and also for scenario development that caters for predicted climate change effects, for example, the spatial migration of wild fisheries (Roy et al., 2007; Coetzee et al., 2008; Mhlongo et al., 2015). Many of our maps, particularly maps of ecosystem threat status, had uptake in sectoral plans, research and management to support fisheries eco-certification, and in environmental impact assessment. The IUCN is advancing such ecosystem red listing efforts to support assessment and

TABLE 1 | A selection of online resources for MSP practitioners and researchers interested in multi-objective planning approaches and tools.

Description	Link
Capacity building on ocean research, all Intergovernmental Oceanographic Commission states	https://classroom.oceanteacher.org/
Caribbean Regional Oceanscape Project, Organisation of Eastern Caribbean States	https://www.oecs.org/ogu-resources/summary-of-regional-strategic-environment-social-assessment-for-crop
Center for Ocean Solutions, Stanford University, United States	https://oceansolutions.stanford.edu/
Coastal and Marine Spatial Planning tools, National Oceanic and Atmospheric Administration, United States	https://cmsp.noaa.gov/data-tools/tools.html
Coastal Resilience, Australia, Caribbean, Indonesia, North America, Mexico and Central America	http://coastalresilience.org
Collaborative Planning for our Oceans, Atlantic, Indian and Pacific Oceans	https://www.seasketch.org/
Community hub for Sustainable Ocean Management and Conservation, United States	https://www.openchannels.org/tools/field-tested-tools
Ecosystem-Based Management Tools, Global network of conservation and management practitioners (institutions from Australia, France, Italy, United States among others)	http://www.natureserve.org/conservation-tools/ecosystem-based-management-tools-network
Mapping Ocean Wealth, Australia, Atlantic Coast, United States, Caribbean, Gulf of California, Indonesia, Micronesia	https://oceanwealth.org/
Marine Geospatial Ecology Tools, Global	http://mgel.env.duke.edu/mget
Marine Integrated Planning, Baltic, Adriatic and Black Sea regions	http://www.plancoast.eu/
Marine Plan Partnership, British Columbia, Canada	http://mappocean.org/
Marine Planning Concierge organizes existing technical approaches, information, and tools in a generalized spatial planning framework, Vancouver Island, Belize, Barbados, New England, The Bahamas, Mozambique, California, British Columbia, Canada	http://msp.naturalcapitalproject.org/msp_concierge_master/
Marine Spatial Planning Programme, Africa, Arctic, Asia, Oceania, Europe, Middle East, The Americas	http://msp.ioc-unesco.org/about/msp-at-unesco/
Marine Spatial Planning, Seychelles, Indonesia, Caribbean, Pacific Islands	http://marineplanning.org/
Marine Spatial Platform, Baltic, Black and North Seas, North East Atlantic and Mediterranean Oceans	https://www.msp-platform.eu/
Open Communications for the Ocean, United States	https://www.octogroup.org/
Platform for knowledge exchange and generation and capacity building for sustainable management, Caribbean Sea, Pacific Islands, Atlantic and Indian Ocean	https://bluesolutions.info/
Seychelles Marine Spatial Plan Initiative	https://seymsp.com/
The Global Oceans Regime, Council in Foreign Relations, United States	https://www.cfr.org/report/global-oceans-regime
Tools for understanding marine biodiversity and assessing good Environmental Status, Gulf of Finland, Kattegat, Southern North Sea, Bay of Biscay, Adriatic Sea, Eastern Aegean Sea, Sea of Marmara, and Western open Black Sea	http://www.devotes-project.eu/
United Nations Environment Programme, Global	https://www.unenvironment.org (search for "Marine Spatial Planning"); http://wedocs.unep.org/handle/20.500.11822/22186; https://www.unenvironment.org/nairobiconvention/nairobi-convention

planning (Bland et al., 2016), but South Africa's national systematic approach covering all ecosystem types is novel. The systematic conservation plan that was used to identify focus areas for offshore protected areas is now being advanced into implementation (Sink et al., 2011), and protection in South Africa's continental exclusive economic zone is being advanced from 0.4 to 5%. Key elements to improve uptake of these maps included translation of biodiversity maps into sector-specific maps to support biodiversity mainstreaming, to serve maps through online Biodiversity GIS portals[4], to provide training to map users and finally the establishment of an annual stakeholder forum to support relationship building and information sharing across sectors.

Our recent efforts have focused on new approaches to incorporate ecosystem services into MSP, including the identification of priority marine areas for food security. We found that our simple and powerful conservation and management messages, aligned with government priorities, had greater impact than complex planning products, and greatly enhanced and facilitated decision-making amongst multi-sector stakeholders.

CONCLUSION

Marine spatial planning is a broadly accepted approach by most governments to better manage the sustainable use of ocean space, and indeed has even become a requirement by some public lenders to ensure sustainable economic development in coastal and marine waters (Smith, 2017). Scientists, managers and policy makers make broad calls for its use to better balance competition among marine uses and address a growing list of issues ranging from renewable energy and aquaculture siting to climate change adaptation and Blue Economy. We complement the already-burgeoning field of MSP by synthesizing very timely spatial analytical approaches and lessons learned from our collective experience working to advance the science and practice of MSP around the world. These lessons are not exhaustive and some of the work is still ongoing. We frame each of the case studies around the general utility of its approach, the importance of providing this information right now to the MSP practitioner community and facilitate the translation of these approaches to new planning processes. In particular, through elaboration of interdisciplinary tools, techniques, and approaches developed to inform real-world MSP processes, we demonstrate the key role that such tools can play to achieve multiple objectives in marine space allocation and management. This diverse collection of studies illustrates how these tools can be applied in different social, political, and ecological settings with different spatial planning needs and data and human resource availabilities.

Commonalities that emerged from our studies include issues of process, as well as technical advances. MSP processes should emphasize transparency, the meaningful participation of all stakeholders, the use of the best available scientific and indigenous knowledge, and align with stakeholder visions, economic imperatives and government priorities. Multi-objective MSP tools should support real-world decision making by addressing issues of efficiency, equity/fairness and conflict, and have the ability to scenario-plan, analyze trade-offs and identify win–win solutions, as well as answer the "where" and the "how much." Advances in the incorporation of ecosystem services into MSP are key aspects of the studies presented.

None of the issues addressed in these case studies is unique to those particular contexts, and therefore the approaches presented here should be useful and transferable to other locations and other planning processes. In particular, our collection of approaches and tools demonstrates that multi-objective planning can be undertaken across a gradient of social-ecological complexity, and is not beyond the scope of under-resourced, data-poor regions. Additional resources for MSP practitioners and researchers can be found online (**Table 1**).

[4]http://bgis.sanbi.org/

AUTHOR CONTRIBUTIONS

AL wrote the manuscript with case study inputs from co-authors. AL and NB were the leaders of the project and conceptualized the idea for research. JS provided strong conceptual input into the structure of the manuscript. NB, JS, SW, SL, KS, AJ, and ZK contributed to the manuscript study and provided the input for final editing. RT and HS contributed to the study led by JS and provided the input for final editing.

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Developing High Resolution Baseline Coast Resource Maps Using World View 2 Imagery for a Coastal Village in Fiji

Ashneel Ajay Singh[1], Anish Maharaj[1], Michelle Kumar[1], Priyatma Singh[1], Sanjay Singh[1], Frank E. Muller-Karger[2], Matthew McCarthy[2], Lionel Joseph[1], Herve Damlamian[3] and Zulfikar Begg[3]*

[1] Department of Science, The University of Fiji, Lautoka, Fiji, [2] College of Marine Science, University of South Florida, Tampa, FL, United States, [3] Geoscience Division, Pacific Community, Suva, Fiji

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***Correspondence:**
Ashneel Ajay Singh
ajaymsp1@gmail.com

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In Fiji, like most Pacific Island countries, there have been numerous reports of degradation of coastal resources, including adverse changes in abundance and stock distribution of numerous aquatic species associated with the coastal habitat. To develop effective management plans, assessment of existing coastal resources is pertinent. High spatial resolution satellite imagery, combined with geographic information systems allow for efficient and synoptic mapping of coastal resources to provide a baseline for developing effective and improved management plans. The purpose of this study was to develop a baseline habitat map of the intertidal benthic cover in Komave Village, Coral Coast, Sigatoka, Fiji. Resource mapping was based on high resolution (2 m) WorldView-2 imagery. Ground-truthing was attained by means of on-site data logging of the intertidal resources, image capturing and GPS recording. Based on these records, the benthic cover was classified into seven classes: 'coral,' 'algae,' 'brown algae,' 'volcanic rocks,' 'sand and gravel,' 'sea grass,' and 'bare.' Ground referencing points were randomly assigned for either supervised classification training or accuracy assessment. A community participatory research approach was used to conduct interviews to assimilate information on fishing sites and coastal land use activities. This exercise explored the social-ecological approach in natural resource management and how it can become an important tool in coastal conservation practices. The coastal resource map generated through this study serves as a baseline for monitoring the status and spatial distribution of the coastal resources in Komave. Annual mapping of the resources and enrichment of maps along with iterative village consultation will enable managers to develop and gauge the effectiveness of coastal management plans. This high resolution map is particularly relevant to Fiji as it is the first of its kind for the country. This work also serves to reduce the global information gap of coastal resource status for Fiji.

Keywords: GIS, coastal resource, benthic cover, Fiji, WorldView-2, satellite imagery

INTRODUCTION

Changes in climatic conditions and anthropogenic influences are negatively affecting coastal ecosystems and threatening resource availability and food security particularly in developing countries of the globe (Barbier et al., 2011; Rice and Garcia, 2011; Williams et al., 2016). South Pacific Island countries (PICs) including Fiji rely heavily on resources from coastal habitats. Increasing human population, amplified harvesting of coastal resources and land use activities exacerbate the problems of climate variability (rain, wind, temperature, cloud cover, sea level, acidification, and natural disaster frequency among others). Coastal monitoring and coastal research studies conducted previously in Fiji have not been designed to evaluate the difference between impacts due to climate change or due to direct human uses (Mimura, 1999; Ellison, 2000; Moreno and Becken, 2009; Le Cornu et al., 2017). A systematic long-term monitoring program of critical sites in conjunction with existing research-based monitoring would improve the identification of effects that different variables have on mangroves, reefs, and benthic communities in Fiji. This is critical for sustainable management practices of coastal resources (Pandolfi et al., 2003; Wilkinson, 2004). This is essential for developing climate resilience for coastal Pacific communities that depend on natural resources and ecosystem services for their livelihood.

Coastal benthic habitats, such as seaweeds, algae, and coral reefs, have high ecosystem-service value. Such habitats and systems act as carbon sinks, and provide coastal inundation protection, wave energy regulation and nurseries for various aquatic animals (Galparsoro et al., 2014). There is evidence that corals are degrading rapidly and are often used in an indiscriminate manner without consideration of sustainability (Brown et al., 2017; Ruppert et al., 2018). Coastal habitat species have particular optimum conditions by which they survive and thrive. These conditions are dependent upon the interaction of multiple factors (Pakeman et al., 2008). There is also documentation of negative anthropogenic effects on coastal fish abundance and distribution (Sundblad et al., 2011; Sundblad and Bergström, 2014; Ruppert et al., 2018). Global reduction in coastal ecosystems have been documented in a number of studies (Barbier et al., 2011; He et al., 2014; Hernández-Delgado, 2015; Lee et al., 2015; Cloern et al., 2016). Different components of the coastal ecosystems have natural interconnectivity and any change will have effect on different components. Coastal ecosystem degradation has been known to cause significant reduction in viable fisheries, nursery habitats for various marine organisms and filtering capability of various aquatic plants and animals (Worm et al., 2006; Barbier et al., 2011).

It is difficult to effectively encompass the spatially complex heterogeneous distribution of coastal cover with conventional methods (Vanderstraete et al., 2005). Typical field-based study usually covers small and fractionated components of the investigated systems (Hochberg and Atkinson, 2003) and is poorly built to identify spatial alterations over time. GIS is a systematic tool for establishing vital baseline information on the distribution of coastal and aquatic resources. It is widely used in ecosystem management and is particularly effective in creating accurate, high spatial resolution base maps of coastal and shallow-water aquatic resources (Aswani and Lauer, 2006; Friedlander et al., 2007; McCoy et al., 2015; Elliott et al., 2018). Benthic habitat maps, including coral cover maps and maps of coastal land resources also provide critical information needed for the management of coastal ecosystems and are used in numerous research and monitoring activities such as coastal development, fisheries and other resource use, coral reef resiliency, connectivity, sea-level change, climate change and ocean acidification (Le Cornu et al., 2017). High resolution base maps are particularly important research tools for monitoring changes and improving sustainable resource management plans. It is possible to create boundaries for marine protected area's (MPA) without using detailed resource maps as has been the case in the past for several of the PICs. However, these approaches rely on field based assessments which do not represent the benthic cover information efficiently which is a common approach in Fiji. As a result, such MPA set-ups give mixed results (Weeks and Jupiter, 2013). In order to understand the baseline abundance and distribution of resources and how these resources change over time within any MPA or field site, accurate maps of the resources are essential. Accurate baseline benthic cover information is a crucial initial step in determining the spatial characteristics and the status of aquatic resources. This is essential for the planning of informed management plans including those for MPA formation. These maps help visualize and understand resource inventories, connectivity between habitats and resource threats (Aswani and Lauer, 2006; Friedlander et al., 2007; McCoy et al., 2015; Elliott et al., 2018).

Previous benthic cover mapping studies in Fiji and the South Pacific region have been of limited accuracy and have not incorporated coastal use and benthic cover change information from the locals (Roelfsema et al., 2007; Baleilevuka et al., 2014). Local residents and villagers utilize and manage costal resources and possess important historical information on the coastal zone (Weeks and Jupiter, 2013). Such survey based information can be integrated with resource mapping and can provide important information on the present and past resource use and status. The integration of local knowledge and scientific methods for development of benthic maps is essential for effective management measures such as formation of MPA areas as shown in different studies (Aswani and Lauer, 2006; Aswani et al., 2007; Teixeira et al., 2013; Aswani and Lauer, 2014). The intention of this research is not only to create a baseline resource map, but to also demonstrate a mixed methodology for merging scientific information with local information in efforts to create an all-inclusive map that can be readily used to design conservation strategies.

Satellite imagery and *in situ* observation are the basis for complex forecasting models and ecosystem-based management (Dowell and Platt, 2009; Beckage et al., 2011; Sherman et al., 2011; Röckmann et al., 2012). One of the key functionalities of GIS is data integration that further enhances visual and digital resource data and provides a detailed understanding of the multifarious nature of the study site. Remote sensing and GIS methods are quite economical in terms of the resources and time involved (Mumby et al., 1999). This is especially advantageous

for PICs with limited resources. In Fiji, spatial distribution of the coastal marine habitats and species distribution are poorly understood. The Komave watershed area including the coast had been identified as critical and nationally significant (Atherton et al., 2005). Earlier consultations with Komave village revealed that villagers were experiencing reduced harvest of fisheries and coastal resources over the past 10 years. This study was intended to provide Komave village with benthic cover maps and information on anthropogenic activities for formulating management plans and formation of an MPA area. To facilitate this, the key objective of this study was to develop a high spatial resolution baseline habitat map of the intertidal benthic cover and resources and map out the anthropogenic activities in Komave village's coastal habitat, Fiji.

MATERIALS AND METHODS

Study Site

The research was conducted within a shallow-water ecosystem in the Southwestern part of Viti Levu (177° 47′ 48″ E and 177° 50′ 59″ E longitude; 18° 16′ 11″ S and 18° 16′ 11″ S latitude) (**Figure 1**). A tourist hot spot for many decades, this area is known as Coral Coast locally and is renowned for its beautiful beaches and reefs. The study region included four reef systems (*Cakaubalavu*, *Cakau Lekaleka*, *Vatumalawa*, and *Nalumu*).

Field Survey

A field campaign was undertaken to collect ground-reference points (GRPs) for every target habitat class using a Trimble Pro 6T receiver and Juno 3D unit and digital camera with a log sheet to take field notes. Target habitats included coral, algae, brown algae, volcanic rocks, sand and gravel, sea grass, and bare (i.e., the beach area). The survey was carried out from the toe of the beach to the reef crest at low tide from July to November, 2016. Preliminary site visits found that some habitats, especially corals were not uniformly distributed but rather sparsely located throughout the study area. As a result, a survey based on randomized transect samples was not a viable option to capture sufficient GRPs for all classes during a limited field season. Instead, *ad hoc* transects were determined subjectively in the field to capture as much benthic-habitat heterogeneity as possible. At each GRP, GPS position was stored (and later differentially corrected in the lab), a photo taken, and the habitat and photo number recorded on the log sheet.

Spatial autocorrelation of habitats was minimized by taking GRPs points at least 20 m apart (McCarthy and Halls, 2014). GRPs collected were randomly halved. One half was used as the

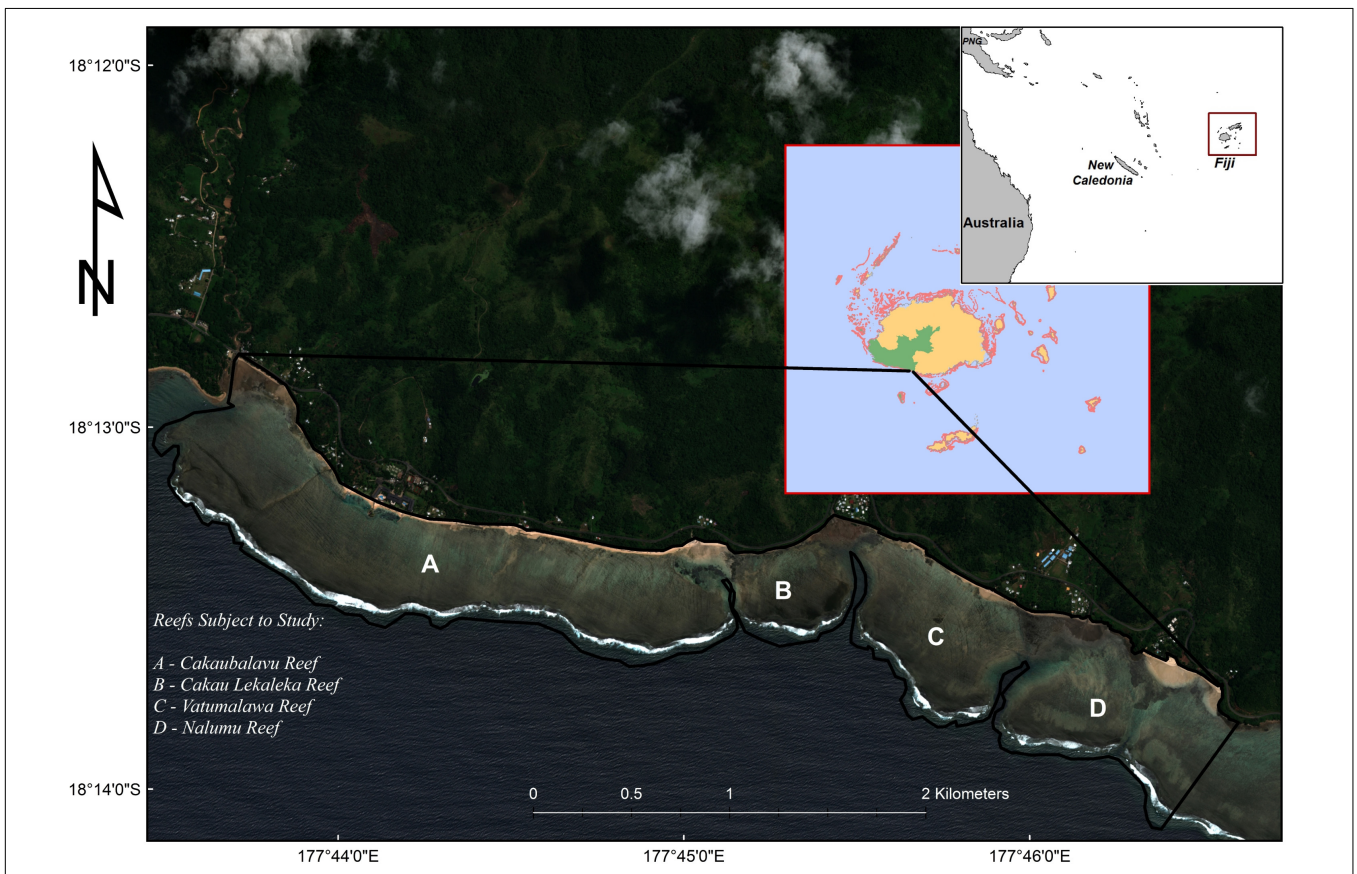


FIGURE 1 | Location of the Komave study site on the southwestern coast of the main Island Viti Levu in Fiji. Komave watershed shown in green in inset of Fiji. Alphabets represent different reefs; (A) *Cakaubalavu*, (B) *Cakau Lekaleka*, (C) *Vatumalawa*, (D) *Nalumu*.

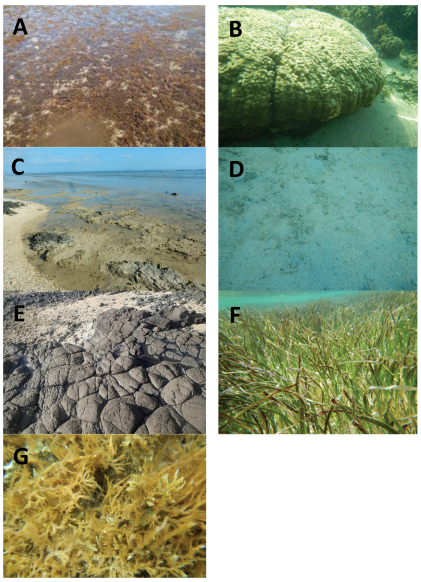


FIGURE 2 | Seven benthic class types based on data collection at Komave. The letters represent each class type: **(A)** brown algae, **(B)** coral, **(C)** bare, **(D)** sand and gravel, **(E)** volcanic rocks, **(F)** sea grass, **(G)** algae.

training sample: a set of data used for classification that is to fit the parameters of the classifier to train the algorithm. The other half was used to validate the classification and compute its accuracy.

Figure 2 shows the seven different types of benthic cover that were targeted for this study. Photos taken at GRPs were analyzed and assessed qualitatively as a quality control check on the initial field determination of habitat. For images with mixed classes, classification was done based on the dominant benthic class.

Imagery Used

A single WorldView-2 image of the study site was used. The WorldView-2 satellite sensor was launched in 2009 and is operated by DigitalGlobe^{TM}. It collects data in eight multispectral bands of visible and near-infrared at nominal spatial resolution of two meters (Digital Globe, 2009). The image was taken on the 24th of January, 2013. It was acquired through a collaborative partnership with the Institute of Marine Remote Sensing, University of South Florida. The data was delivered as a "LV1B" (i.e., sensor and radiometrically corrected only; Cheng and Chaapel, 2010).

Image Pre-processing

ENVI's WorldView Radiance tool was used for radiometric calibration. This produced the radiance values by multiplying the metadata-based gain with the pixel value and adding the offset (Harris Geospatial Solution, 2016). Fast Line-of-sight Atmospheric Analysis (FLAASH) was used to radiometrically correct the image. FLAASH reduces the atmospheric effects on the imagery and produces at-the-surface reflectance (Phinn et al., 2012). Geometric corrections were performed for correct alignment of GRPs to the imagery. To maximize the computational capacity, the study site was cropped out of the image. Some areas were masked out of the image to avoid misclassification with target habitats (e.g., "white water" areas of the reef crest).

Supervised Classification and Accuracy Assessment

This classification is based on the assumption that multispectral satellite imagery measures different spectral signatures for different benthic cover. Spectral signatures describe how objects reflect solar radiation and correspond to physical and biological characteristics.

Maximum Likelihood Classifier (MLC) was chosen to carry out the classification since it is the most widely used supervised classification method (Yang et al., 2015). MLC automatically categorizes pixels in an image into a trained (i.e., target) class (Vahtmäe et al., 2012). MLC evaluates the brightness of one band compared to the other (variance and covariance) in the training class and then it categorizes pixels based on its maximum probability of belonging in a class (McCarthy and Halls, 2014).

Quantitative assessment of how accurately the pixels have been grouped into the user-defined classes is very important (Ismail and Jusoff, 2008). This is achieved by conducting an accuracy assessment on the classified image. The remaining half of the GRPs were used to construct a confusion matrix in ENVI. Using the results of the confusion matrix, the accuracy of the map was determined.

Derived Bathymetry

Bathymetric maps are important for understanding more about climate change effects. This type of survey can be used to alert scientists on ongoing and potential impacts including beach erosion, sea level rise and land sinking. Landsat 8 imagery was used to derive the bathymetry profile. The image comes with a coastal band where the wavelength penetrates deeper into the water column than the common blue band. The LANDSAT8 imagery was selected based on three criteria; (1) High sun elevation (ideally greater than 60 degree); (2) High tide to maximize submerged area over the reef flat; (3) Cloud free ratio. The processing was undertaken in Python2.7 for easy sharing and enhancement of the codes used for deriving bathymetry.

Climate Change Perception and Community Participatory Exercise

A qualitative, in-depth individual interview was also conducted at Komave village. The village consists of about 40 households with a population of around 208 individuals with approximately 114 females and 94 males. A total of 40 participations were interviewed between the ages of 25–45 consisting of 18 males (45%) and 22 females (55%). A qualitative research approach was particularly important to ascertain the coastal activities and climate change perception of the villagers. The interviews comprised of 'yes,' 'no,' and 'no change' type responses on fisheries resources, socioeconomic livelihood, climate change perception, tourism and benthic cover change. A participatory action research approach was also employed to fine tune the maps

produced. Large color printed maps were provided to the villagers and human use activities were marked by the participants for different aspects including, fishing sites, proposed developments, picnic spots and marine protected areas (MPAs). These maps were later scanned, georeferenced to their real world coordinates and the different human use categories digitized into different layers. These sites were further verified by the villagers physically identifying the different sites and taking of GPS points. This information was used for adding GIS layers for local knowledge to the maps. Fisheries resource information was gathered as the villagers are reliant on a subsistence lifestyle and fisheries harvest for their socioeconomic livelihood.

For assurance of ethical considerations, the research permit was obtained from the Ministry of iTaukei[1] Affairs (MIA), Ministry of Fishereis and the Ministry of Education, Fiji. The MIA looks after the affairs of the *qoliqoli*[2] and other locally owned resources through Provincial administration. Each visit to the site was accompanied by provincial council members and village representatives. Sharing and publication of information on the status and use of *qoliqoli* resources was part of the project and an agreement among the villagers, MIA, The University of Fiji and the donor agency [United States Agency for International Development (USAID)]. Village representatives formed part of data gathering exercise and villagers were continuously updated on the progress of the project and data was shared with them including all the maps that were generated.

RESULTS

The habitat map results, including each of the seven target habitats are shown in **Figure 3** along with the site's bathymetry map. Bathymetry map shows the study site to be quite shallow with most part of the study area below nine meters.

The confusion matrix of the seven benthic habitat types is represented in **Table 1**. The matrix shows an overall accuracy of 91.6% with a kappa hat classification value of 0.9. Kappa hat classification is the measure of agreement between the classification map and the reference data. A kappa of 0.8 or above is considered a good classification and a kappa of 0.4 or below is considered a poor classification. Algae were the most dominant habitat type. The most dominant biotic cover was algae followed by sea grass, brown algae, and coral (**Table 1**).

Data gathered from semi-constructed questionnaires are summarized in **Table 2**. Historical fisheries data for Komave area was not available. The questionnaires formed an alternative method of gauging to some extent the past and present situation of fisheries status. The villagers have heavy reliance on access to fishery resources for their socio-economic livelihood. Although it is not quantitative, the data does provide anecdotal indication of fisheries decline over the past 10 years. It can also be seen that most of the interviewed personnel associate the decline in fisheries status with climate change and not land use activities even though there has been no scientific justification yet for the site. Information was also gathered on various aspects including participant's observation of benthic cover change over time. **Figure 4** shows the overlay of human activities over the benthic cover map. Proposed developments, fishing sites, picnic spots, and MPAs are shown. Also shown are the locations of two beach hotels within the Komave coast. A list of all the GIS layers produced is shown in **Table 3**. Interactive maps and more information is available in the **Supplementary Material**.

DISCUSSION

Habitat maps provide important information on the distribution and status of natural resources. They provide effective and reliant baseline information for appropriately gauging local management and economic options. Activities such as tourism recreational areas can be planned out based on the attractive natural resources such as coral reefs. Frequently generated maps of the same area can be used as a monitoring tool to keep track of the effectiveness of different management options. Impact of natural disasters, environmental regime shifts and extreme events can be gauged and monitored.

Critically analyzed benthic habitat maps can inform and improve the design of resource assessment surveys, which are routinely conducted to site and evaluate the potential impacts of development projects. Naidu et al. (2018) mapped out five classes of benthic resources in Komave with an overall accuracy of 71%. In this study a total of seven classes were mapped out with an overall accuracy of 91%. These maps can be used for marine and freshwater inventory surveys such as monitoring fish abundance and diversity, monitoring of the coral reef ecosystem, socio-economic analysis of the community, design and evaluation of MPAs and design and monitoring of fisheries management plans. The setting up of marine reserves and proper monitoring have the potential to support mitigation of, and adaptation to, climate change (Roberts et al., 2017). Monitoring of aquatic habitat alteration due to climate change and land use activities can be effectively done over time and targeted management policies can be effectively formulated such as community action plans toward conservation and sustainable management of resources.

The results of benthic cover mapping in **Figure 3** serve as baseline information but it does not show changes in the cover over time. The cover change information is shown in **Table 2** using community participatory approach. The information is qualitative but is does inform how the benthic cover has generally changed over time. Over a period of 10 years, it is noted that coral and sea grass cover has reduced, brown algae has seen no significant change while green algae cover has significantly increased. These are indications that the coastal resources have seen a decline in the past decade. The changes noted can partially be attributed to anthropogenic activities in Komave. Location of hotels, picnic spots and proposed developments are observed to be in close proximity to the coast (**Figure 4**). It is evident from **Table 2** that the fisheries resources in Komave are facing issues of declining fish sizes and movement of targeted fish stock further away from the shore. This has significant repercussions on catch and costs associated with reaching the fishing sites. With time it

[1] Refers to native Fijians.

[2] Traditional Fijian term for natural coastal resources that are owned by locals.

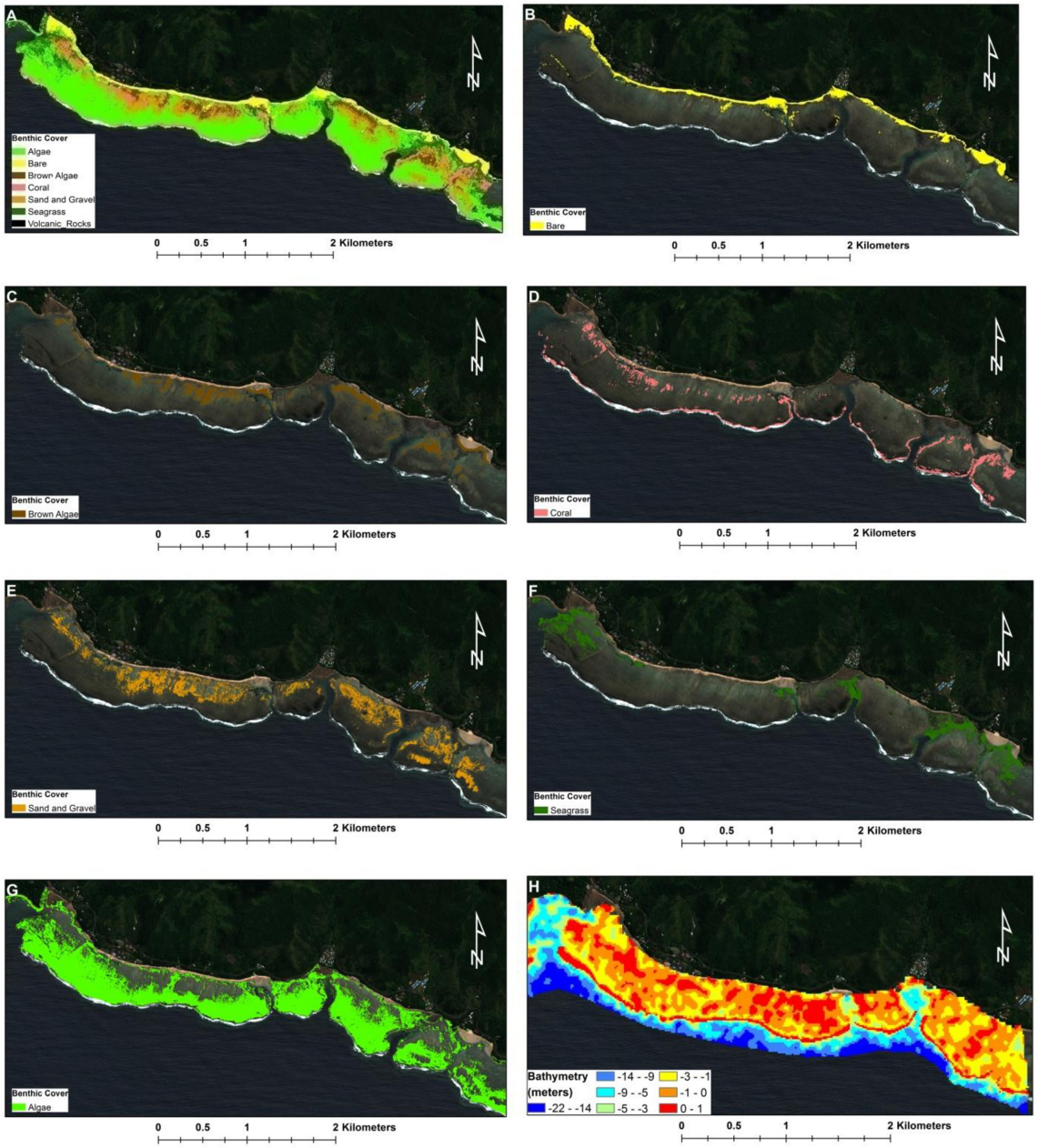


FIGURE 3 | Benthic cover map for Komave coastal habitats. **(A)** Total benthic cover map with all seven classes represented, **(B)** bare, **(C)** brown algae, **(D)** coral, **(E)** sand and gravel, **(F)** sea grass, **(G)** algae, **(H)** is the bathymetry[3] map showing the depth distribution of the study site.

costs fishermen more to catch less. Changes in sufficient access to fisheries resources have had significant negative impact on the socioeconomic livelihood of most of the households in the village.

Figure 3 and **Table 1** show excessive benthic algal cover in the area and **Figure 4** shows anthropogenic activities in the area including tourism and fishing activities. Elevated levels of algal growth have been shown to be strong indication of heavy metal pollution due to anthropogenic activities (Chakraborty et al., 2014; Phillips, 2017). The study conducted by Mosley and Aalbersberg (2003) at Komave, revealed overgrowth of algae due to increased levels of nitrates and phosphates in sea water. Elevated nutrient level resulted in loss of fish and invertebrate biodiversity as a loss of habitat heterogeneity. As a result, fishermen made use of small petroleum based engine boats to move further offshore for the catch. The combustion and possible leakage of petroleum fuel led to release of volatile and organic pollutants and heavy metals in sea water. In addition to this, waste water discharge from hotel

[3]Note: Bathymetry map is not to be used for navigational purposes.

TABLE 1 | Confusion matrix results and accuracy assessment for benthic cover maps at Komave.

Class	Producers accuracy	Users accuracy	Overall accuracy	Kappa coefficient	Percentage cover (%)
Algae	100	50.43	91.6	0.9	47.62
Coral	96.15	97.40			7.48
Bare	78.85	98.80			8.12
Volcanic rock	80.89	100			0.25
Sea grass	97.5	98.73			9.76
Brown algae	93.86	100			7.73
Sand and gravel	100	95.42			19.03

Also shown are the spatial coverage percentages for the different benthic habitat types.

TABLE 2 | Local knowledge information through community participatory approach on fisheries resources, socioeconomic livelihood, climate change perception, tourism and benthic cover change for Komave village.

Question	Yes (%)	No (%)	No change (%)	Total respondents
Has the number of fish caught per fishing trip declined over the past 10 years?	98	0	2	40
On average has the catch size of your targeted species declined in the past 10 years?	91	5	4	40
Has your fishing site moved further offshore in the past 10 years?	96	1	3	40
Has the cost of reaching your fishing site increased over time?	99	0	1	40
Has your financial gains been negatively affected over the years from harvest of adequate fisheries resources?	100	0	0	40
Can your socioeconomic livelihood survive without harvesting enough fish?	2	93	5	40
Do you think the change in fish abundance and distribution is due to climate change?	100	0	0	40
Do you think the changes in fish abundance and distribution due to land use activities?	2	98	0	40
Has the abundance of live corals significantly reduced in the coastal area over the last 10 years?	90	3	7	40
Has the abundance of brown algae significantly reduced in the coastal area over the last 10 years?	3	2	95	40
Has the abundance of green algae significantly increased in the coastal area over the last 10 years?	100	0	0	40
Has the abundance of sea grass significantly reduced in the coastal area over the last 10 years?	91	0	9	40
Does the village have any authority for managing tourism recreational activities in the coastal area?	0	100	–	40
Do you think tourism activities are being harmful to the coastal resources?	20	80	–	40

and improper waste disposal and natural and anthropogenic activities have significantly contributed to heavy metal discharge in sea water which impairs physiological processes such as fertilization in corals.

Joseph et al. (2019) carried out a study in Komave coast on sea grass abundance and nitrate levels for 2 years between 2016 and 2017. A reduction in cover of five different sea grass species was noted. A significant negative relationship was determined between nitrate levels and total sea grass cover. Mosley and Aalbersberg (2003) reported that the nitrate and phosphate levels at the west of Komave village were 2.12 and 0.15 μM respectively during the beginning of April 2002 and 0.98 and 0.12 μM during the end of June 2002. The mean nitrate level for the water samples at the site was 1.55 μM and the mean phosphate level was 0.135 μM. The mean nitrate and phosphate levels exceeded levels considered to be harmful to coral reef ecosystems (nitrate > 1.0 mM; phosphate > 0.1 mM). Increased nutrient level has significantly increased the growth of phytoplankton and other algae (*Sargassum* sp.) which form the base of ocean food chain. Moreover, elevated levels of nutrients in coral reef ecosystems in Fiji has been shown to shift species dominance of the slow growing coral reef building (stony calcifying) corals to a larger non-calcifying faster growing corals. This process has resulted in mortality and loss of biodiversity of live corals and loss of settlement sites for coral larvae (Mosley and Aalbersberg, 2003). Increased levels of inorganic phosphorus led to a reduction in density of stony corals causing them to crumble and lose their strength. Elevated levels of Nitrate concentration over the study period indicated possible nutrient run off from anthropogenic and farming activities (crops and livestock) along the huge watershed flow leading to Komave coast (Joseph et al., 2019). With this knowledge, pressing issues such as the declining coastal fisheries resources can be better understood and managed.

A community-based participatory approach was adopted to engage the Komave villagers in the mapping process. The community members consisting of both men and women verified the maps and pointed out important fishing and gleaning zones, MPA areas, picnic spots and proposed development sites. The map was modified based on the information provided by the community members. Corrections were made accordingly in the map produced. Community-based participatory approach turned out to be an important prerequisite to effective data analysis and decision-making. It is particularly important because in several cases, community responses to climate change involve management actions that aim to protect coastal areas (Wiber et al., 2004). The local valuation of traditional ecological knowledge, the familiarity with resource use and distribution, and a sense of ownership of coastal reserves are essential aspects of community social structure that could be creatively integrated with scientific objectives to bring about societal transformations needed for conservation of the coastal resources. In this project, the community-based participants were not merely seen as external stakeholders, but as equal partners with local knowledge that was utilized to evaluate the research findings. Irrefutably, participatory methods can support informed and coordinated decision-making for marine resources. The findings of this study call for a comprehensive

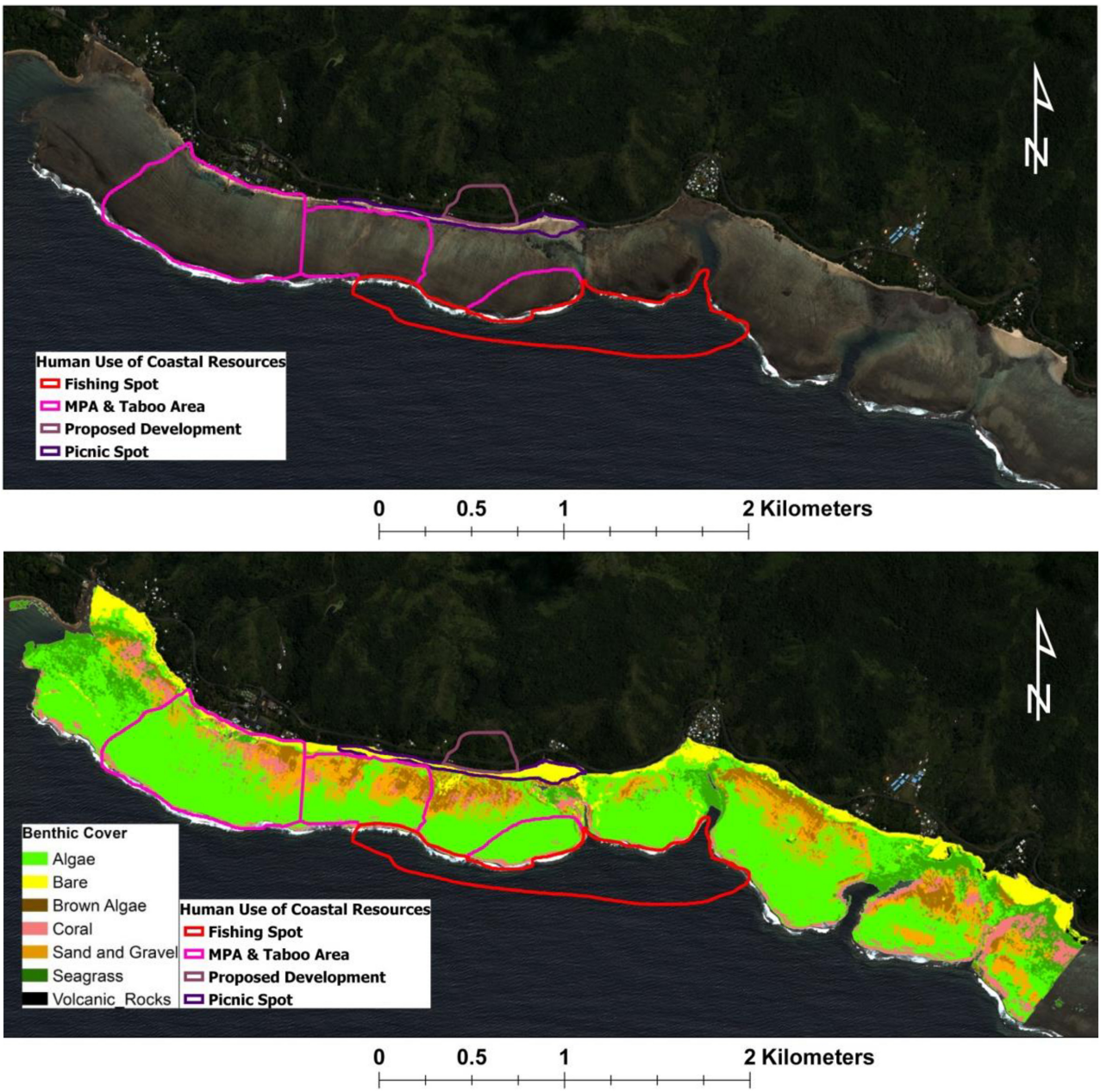


FIGURE 4 | Benthic cover map for Komave coastal habitats overlaid with human use activities; fishing sites, proposed developments, picnic spot and marine protected area (MPA)/taboo area. **(Top)** Without benthic cover classes; **(Bottom)** with benthic cover classes.

method that characterizes stakeholder practices as they relate to specific parts of the ocean using GIS and mapping support technologies. Using an all-inclusive integrated system, authors argue that managers can create meaningful participation for marine spatial planning, but there is an urgent need to overcome the existing complexities in engaging various marine resource users by increasing stakeholder participation and empowerment. It is difficult to understand marine conservation issues through the lens of one discipline only. While the core of this research involved creating high resolution baseline coastal resource maps, the *talanoa*[4] sessions with the community participants provided critical information about the resource use and distribution that gave more insights into the conservation issues. Such integrated approaches that include both social and natural sciences are proving to be an effective tool for marine conservation and management, particularly in traditional coastal villages that rely on ocean resources for their livelihood (Leenhardt et al., 2015).

[4]Indigenous Fijian term that describes informal discussions.

TABLE 3 | GIS layers for benthic habitat and local knowledge for Komave coastal region.

GIS layer	Description
Bare	Quadrants without any features between shoreline and mainland vegetation.
Brown algae	Areas covered predominantly with brown algae.
Coral	Areas covered predominantly with live corals. All live species of soft and hard corals are included.
Sand and gravel	Areas covered predominantly with sand and/or gravel in the intertidal zone.
Sea grass	Areas covered predominantly with sea grass. All species of sea grass are included.
Algae	Areas covered predominantly with green algae.
Volcanic rocks	Areas covered predominantly with volcanic rock outcrops.
Bathymetry	Depth distribution of the study site aquatic zone is shown.
Fishing sites	Fishing areas identified by villagers through participatory exercises.
MPA/Taboo area	Restricted and protected harvest areas identified by villagers through participatory exercises.
Picnic spot	Recreational areas for tourists and locals identified by villagers through participatory exercises.
Proposed developments	Areas of proposed future developments identified by villagers through participatory exercises.

The maps generated on the seven benthic classes in this study will enable decision makers and stakeholders to use these as reference material for the state of the coastal resources. Annual mapping will allow for monitoring and change detection in the abundance and distribution of the benthic classes. The socioeconomic survey provides a useful tool for filling in data gaps and gauging general change patterns in coastal resources. In the South Pacific, most developing states have very little to no data availability on coastal resources. As is the case for Komave, a combination of benthic cover mapping and local knowledge integration can allow for meaningful baseline information for development of sustainable management plans. The procedure used can be easily replicated for different parts of Fiji as well as other PICs. Such baseline information is essential to gauge the coastal resource status in the PICs region and inform the development of regional management plans. In a global context, coastal resource information for PICs is lacking in many instances and creates information gaps for developing targeted global management plans. The kind of study done here can be used to fill in such data gaps.

Villagers' perceptions attribute coastal resource changes to the changing climate (**Table 1**). Upon further query, all respondents advised that they were able to observe changes in the seasons and local media explains the impacts of climate change as such. This study showed to some extent that anthropogenic activities can have significant negative impact on coastal resources. Anthropogenic actions are much more feasible to control compared to the impacts associated with the global phenomenon of climate change. The maps and associated databases generated from the project will enable stakeholders such as government departments, research institutions and universities to carry out further interdisciplinary studies. This includes probable impacts of coral reef geomorphology due to resource use or ocean acidification in response to climate change. Changes in aquatic biodiversity due to expansion or reduction of habitat of different species as a result of ocean warming, sea level change or changes in ocean pH can be monitored and integrated with maps and databases from other regions generated using the same satellites. This will enable researchers to monitor and model global changes in oceanic properties related to climate change such as comparative sea temperature and pH levels.

CONCLUSION

The maps generated through this project are intended to assist the villagers in the identification of critical areas and the setting up of marine reserves. The integration of local knowledge with benthic cover maps provides improved tools and information for resource management. Different layers provide varied information from land-use and fishing activity to the layout of benthic habitats. Community participatory exercises and regular gathering of benthic cover information will add to the resource maps and possibly show changes over time. These changes can be monitored and managed with management effectiveness can be tracked over time. Policy makers can use the information to improve adaptation strategies currently in place and increase the resilience of communities toward the impacts of climate change leading to a greater food security and enhanced livelihoods.

ETHICS STATEMENT

For assurance of ethical considerations, the research permit approving this study was obtained from the Ministry of iTaukei Affairs (MIA), Ministry of Fishereis and the Ministry of Education, Fiji. The MIA looks after the affairs of the qoliqoli and other locally owned resources through Provincial administration. Each visit to the site was accompanied by provincial council members and village representatives. Sharing and publication of information on the status and use of qoliqoli resources was part of the project and an agreement among the villagers, MIA, The University of Fiji and the donor agency [United States Agency for International Development (USAID)]. All participants gave verbal informed consent to participate.

AUTHOR CONTRIBUTIONS

AS assisted in field data collection and verification. He lead the manuscript write-up. AM lead the data collection, analysis, verification, and mapping exercise. MK assisted in data collection, verification, and analysis. PS assisted with data collection, verification, analysis, and assisted with the write-up. SS contributed to data analysis and manuscript write-up. FM-K motored and guided the methodology, data collection, and analysis. MM assisted in methodology development, data analysis, and manuscript editing. LJ assisted in data collection and analysis. HD and ZB contributed to field data collection, verification, and accuracy assessment.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00207/full#supplementary-material

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Improving Representation of Human Well-Being and Cultural Importance in Conceptualizing the West Hawai'i Ecosystem

Kirsten M. Leong[1], Supin Wongbusarakum[1,2], Rebecca J. Ingram[1,2], Alexander Mawyer[3] and Melissa R. Poe[4]*

[1] Ocean Synthesis and Human Dimensions Program, Ecosystem Sciences Division, Pacific Islands Fisheries Science Center, Honolulu, HI, United States, [2] Joint Institute for Marine and Atmospheric Research, University of Hawai'i, Honolulu, HI, United States, [3] The Center for Pacific Islands Studies, University of Hawai'i, Honolulu, HI, United States, [4] Washington Sea Grant, University of Washington, Seattle, WA, United States

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***Correspondence:**
Kirsten M. Leong
kirsten.leong@noaa.gov

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Ecosystem-based management approaches are increasingly used to address the critical linkages between human and biophysical systems. Yet, many of the social-ecological systems (SES) frameworks typically used in coastal and marine management neither represent the social and ecological aspects of the system in equal breadth or depth, nor do they adequately operationalize the social, or human, dimensions. The National Oceanic and Atmospheric Administration's West Hawai'i Integrated Ecosystem Assessment, a program grounded in ecosystem-based management, recognizes the importance of place-based human dimensions in coastal and marine resource management that speak to a fuller range of social and cultural dimensions of ecosystem-based management. Previous work with stakeholders in West Hawai'i revealed noteworthy SES dynamics and highlighted both the importance and lack of understanding of the links between ecosystem services and human well-being, particularly services that enhance and maintain active cultural connections to a place. While cultural ecosystem services and human well-being are often recognized as important elements of SES, there have been substantial barriers to fully representing them, likely due to perceived difficulties of measuring non-material benefits and values, many of which are socially constructed and subjective. This study examined SES frameworks related to cultural ecosystem services and human well-being to advance the representation and operationalization of these important concepts in coastal and marine management. We describe key insights and questions focused on: (1) points of inclusion for human dimensions in SES models, (2) culturally relevant domains of human well-being and related indicators, (3) the importance of place and its interaction with scale, and finally (4) the tension between a gestalt vs. discrete approach to modeling, assessing, and sustainably managing social-ecological systems.

Keywords: cultural ecosystem services, human well-being, ecosystem-based management, social-ecological system, integrated ecosystem assessment, West Hawai'i, coastal management, marine management

INTRODUCTION

Ecosystem-based management has gained broad recognition as a crucial means to improve conservation and sustainable use of marine systems, through coordinated management of cumulative impacts from multiple sectors (Mcleod et al., 2005; Leslie and Mcleod, 2007). This approach has been embraced by the United States National Oceanic and Atmospheric Administration, the agency responsible for the stewardship of the nation's ocean resources and their diverse habitats. Over the past decade, the conventional focus by the National Marine Fisheries Service (NMFS) on single species fisheries management has broadened to an ecosystem-based approach that includes multiple fisheries and multiple sectors aside from fisheries, such as tourism, coastal development, and marine-related industries (National Marine Fisheries Service, 2016), and interactions between and within biophysical, social, and economic systems (Link, 2010). Ecosystem-based fisheries management (EBFM) adopted by NMFS, thus applies an ecosystem approach to managing fisheries with a focus on multiple biophysical and socioeconomic objectives, with growing concern with procedural equity and the distribution of ecosystem benefits and services (Levin et al., 2018).

The Integrated Ecosystem Assessment (IEA) program was established in 2009 as one tool to help the agency move toward ecosystem-based management. IEAs have focused on large marine ecosystems, with the primary objective to provide a sound scientific basis for ecosystem-based management by synthesizing and providing "[. . .] analysis of information on relevant physical, chemical, ecological, and human processes in relation to specified management objectives" (Levin et al., 2008, 2009). Levin et al. (2016) provides an overview of the progress the IEA program has made toward viewing ecosystems through a coupled social-ecological systems (SES) lens that explicitly includes the "social" elements of SES, or human dimensions (including social, economic, and cultural), in evaluating ecosystem status, risk, and trade-offs of management alternatives to sustain human well-being. IEAs initially followed the established Driver, Pressure, State, Impact, Response (DPSIR) approach to ecosystem assessments (as described by Kristensen, 2004), which largely includes humans only via activities that put negative pressure on the biophysical ecosystem. IEAs then broadened to include benefits to humans via ecosystem services, as a Driver, Pressure, State, Ecosystem service, and Response (EBM-DPSER) model (Kelble et al., 2013). Recent work draws on a more holistic SES approach, conceptualizing the biophysical environment and human dimensions of the system as interconnected, influenced by both biological and social drivers, mediated by habitat and local social systems, affecting ecological integrity and human well-being, and often linked through human activities (Levin et al., 2016; Karnauskas et al., 2017; Ingram et al., 2018).

Developing measurable indicators for the human dimensions of SES has been challenging. Reviews of frameworks designed for broad ecosystem application have noted that most: represent the social and ecological systems in unequal breadth or depth; ambiguously operationalize social concepts (Binder et al., 2013; Hinkel et al., 2015); or draw unevenly from the range of social sciences (Fabinyi et al., 2014). In addition, available social data at the scale of large marine ecosystems is usually limited to information about population demographics, methods and patterns of resource use, and economic performance, which do not adequately capture important linkages between biophysical conditions and cultural benefits of nature (Daniel et al., 2012). This has been a challenge for SES work in general. Kittinger et al. (2012) noted that far more attention has been paid to understanding biophysical dynamics than human dimensions of coral reef management and that there are limited efforts that link social information to biophysical conditions, a concern echoed more broadly by Rissman and Gillon (2017). Our research focuses on the desired outcomes of coastal and marine management related to particular human dimensions of management, namely human well-being and related ecosystem services, to advance their representation and operationalization in SES frameworks for coastal and marine management. Future work will build on these frameworks to identify specific indicators. In addition to contributing to theory and methods, this effort will improve annual Ecosystem Status Reports, which summarize the status and trends of IEA SES components, and will allow better evaluation of the success of management interventions with respect to desired human well-being outcomes.

Many SES frameworks take an anthropocentric perspective, viewing the ecological system as a provider of ecosystem services that support human well-being (Binder et al., 2013; Kelble et al., 2013; Partelow and Winkler, 2016), and often draw on the four categories of ecosystem services described in the Millennium Ecosystem Assessment (2005). Three of those four categories, provisioning (e.g., food and water), regulating (e.g., climate and flood regulation), and supporting (e.g., nutrient cycling) services, can be quantified through well-established methods and incorporated into these types of assessments and management, while the fourth, cultural ecosystem services (e.g., aesthetic, spiritual, recreational experiences), continues to require significant conceptual, methodological, and empirical attention (Daniel et al., 2012; Hernández-Morcillo et al., 2013; Pascua et al., 2017). Cultural ecosystem services are not limited to indigenous or traditional cultures, but rather refer to the often intangible or non-material benefits derived through people's relationship with an ecosystem, evidenced in their spiritual values, social interactions, and emotional experiences (Millennium Ecosystem Assessment, 2005; Chan et al., 2011; Small et al., 2017). Cultural ecosystem services can contribute to a person's well-being via processes such as fostering and maintaining connections to place, identity, values, or directly enabling cultural practices (Chan et al., 2012; Fish et al., 2016; Poe et al., 2016; Pascua et al., 2017), which in turn can affect how people interact with the ecological system. There have been substantial barriers to operationalizing cultural ecosystem services in ecosystem service frameworks for coastal and marine management. One barrier is the predominant focus on uni-directional flows of ecosystem goods and services, which has become institutionalized in the Open Standards for the Practice of Conservation (The Conservation Measures Partnership, 2013). This approach does

not require examination of the social system (including aspects of resource stewardship and governance), as it presumes that the state of the biophysical system automatically determines the ecosystem services received, rather than feedbacks between SES (Chan et al., 2012). Other barriers are related to the perceived difficulties of measuring non-material benefits that are socially constructed and subjective rather than material components of the ecosystem (Daniel et al., 2012; Fish et al., 2016), and a resistance by decision-makers to drawing on anthropology and related qualitative social sciences to understand non-material cultural dimensions (Bennett, 2019). In response, multiple social science approaches have been identified to help improve the robustness of cultural ecosystem services indicators, although they are not regularly implemented in practice, emphasizing the importance of including multiple social science traditions on transdisciplinary teams for comprehensive SES assessments (Daniel et al., 2012; Fish et al., 2016; Small et al., 2017).

A growing body of literature also has focused more directly on development of human well-being indicators for ecosystem assessment and management (e.g., see Dillard et al., 2013; Wongbusarakum et al., 2014; Breslow et al., 2016). These efforts define human well-being as "people's ability to live a life they value" (Wongbusarakum et al., 2014, p. 4) and as, "a state of being with others and the environment, which arises when human needs are met, when individuals and communities can act meaningfully to pursue their goals, and when individuals and communities enjoy a satisfactory quality of life" (Breslow et al., 2016, p. 251). Attention is paid to a range of well-being domains including livelihoods, health, education, and governance. Common to these efforts is the focus on the meanings people place on their interactions with the environment and society, and their abilities to act and enjoy their lives. Yet, as with ecosystem services assessments, material aspects of well-being are predominantly measured, while non-material elements (such as sense of place, cultural values, and identities) are lacking (Mckinnon et al., 2016), and may require additional social scientific methods to develop appropriate metrics and next-generational conceptual models, especially those designed to examine subjective perceptions of well-being (Breslow et al., 2016, 2017).

Both cultural ecosystem services and human well-being approaches to natural resource management recommend developing place-based indicators tailored to management needs due to the relational nature of environmental spaces, natural resources, cultural practices, and perceived goods and benefits (Dillard et al., 2013; Breslow et al., 2016; Fish et al., 2016; Partelow and Winkler, 2016). Unlike other NMFS IEAs, which span geographic areas as large as the Gulf of Mexico, or the entire west coast from Washington state to Baja California (known as the California Current), the West Hawai'i IEA focuses on a smaller area where there has been a history of marine conservation activity. It encompasses the western coastal and marine ecosystems off Hawai'i Island, with the western boundary dictated by ecology linked to West Hawai'i and land based processes and activities included to the extent they affect marine ecosystems (**Figure 1**). This limited geography makes it conducive for exploring place-based ecosystem assessments. In addition, previous work with stakeholders in West Hawai'i identified cultural ecosystem services as exceptionally vulnerable to ecosystem change and an area that needed to be examined in greater detail to ensure human well-being (Ingram et al., 2018). These conditions also indicate the utility of a biocultural

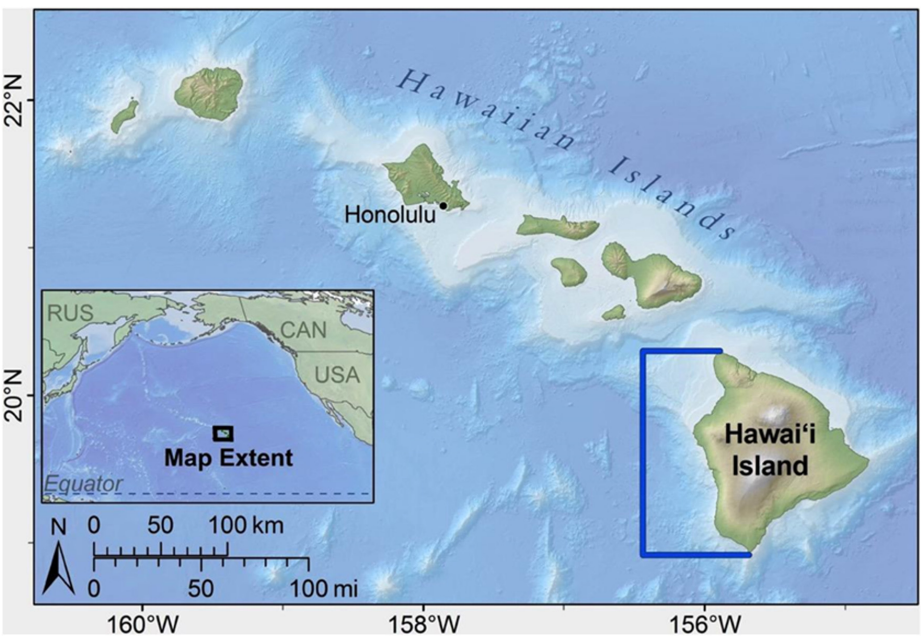


FIGURE 1 | Approximate geographic extent (blue line) of the West Hawai'i Integrated Ecosystem Assessment. Source: Joey Lecky, Pacific Islands Fisheries Science Center, and NMFS.

approach which emphasizes linkages between biophysical and sociocultural components of SES, partners with local communities to identify feedbacks between ecosystems and human well-being, and relies on multiple-knowledge systems to identify management interventions that can meet objectives of stakeholders with diverse priorities and worldviews (Gavin et al., 2015; Sterling et al., 2017a; Gavin et al., 2018). Following this type of approach should lead to indicators that are place-based, culturally grounded, and reflective of both human well-being and the resilience of the associated ecosystem (Sterling et al., 2017b).

West Hawai'i is home to a highly productive and diverse marine ecosystem, supporting an abundance of tropical corals, reef fishes, sea turtles, cetaceans, and manta rays (Gove et al., 2016). The marine resources in the region provide a multitude of ecosystem services valuable to people both locally and globally, such as tourism (the Hawai'i Visitor Bureau reports over 1 million visitors in West Hawai'i annually), aquaculture, protection from wave and storm impacts, fishing, and innumerable cultural practices and activities. West Hawai'i also encompasses a complex social and cultural context, with communities featuring: indigenous Kanaka Maoli (Native Hawaiian) families who may possess profound and diverse indigenous ecological knowledge relative to marine, coastal and terrestrial domains and linkages among them; long-established local communities with families rooted in the plantation and labor histories of different agricultural projects from the late 19th century to the present; relatively recently established families primarily from continental North America and Asia; diasporic communities of Pacific Islanders from elsewhere in the region; and, finally, large numbers of transient tourists and the service providers that cater to them in numerous activities across terrestrial and marine domains. The complexity of the social and ecological context, and the small spatial scale, in West Hawai'i affords a unique opportunity to examine how to better integrate social datasets and place-based human well-being metrics into ecosystem-based management of SES and to improve local management.

MATERIALS AND METHODS

Given our research focus and location, we adopted the biocultural approach described above. Fundamental to this approach is respect for the plurality of priorities, worldviews, and governance systems through which stakeholders interact with resources and their management. Thus, we sought project consultants and community partners to help us better understand: how to appropriately integrate human dimensions into the West Hawai'i IEA project; the most relevant potential human well-being indicators to pursue in more depth; and how management could more effectively develop the links between people and the coastal and marine resources in order to achieve a more sustainable outcome that balances ecological and human well-being.

We applied this approach to three activities: input from specialists; synthesis of relevant literature; and qualitative data collection through group discussions and pilot interviews.

Input From Subject Matter Experts, Resource Managers, and Local Community Leaders

We invited a group of mentors from various backgrounds to help guide the development of our project. These included six subject matter experts, two resource managers, and three local knowledge and community leaders. The subject matter experts were identified based on their experience and knowledge working in the following areas: cultural ecosystem services; sense of place; monitoring human well-being in conservation or natural resource management; and research or collaborative work with communities in Hawai'i or indigenous peoples who rely on marine and coastal resources. The two resource managers have years of experience working in West Hawai'i and are involved in day-to-day efforts bridging research, management, and community needs. The three local knowledge and community leaders were recommended by staff of conservation organizations in West Hawai'i; had a strong connection with West Hawai'i; and worked toward sustainable development, conservation of natural and cultural resources, or natural resource management.

Throughout the project, we sought feedback and advice from the mentor group as a whole or approached individuals as needed for their specific areas of expertise. Subject matter experts helped identify relevant sources of literature for review and provided input on our study design, data collection protocols, and methods. Managers and community leaders identified ways that IEA research can contribute to management and community needs, helped us identify communities that might benefit from this type of work, and helped build relationships with these communities who subsequently continue to partner in the research.

In addition to the project mentors, we involved over a dozen West Hawai'i and Hawai'i State resource managers in multiple ways. We discussed current and future goals of the management agency and identified gaps that should be filled; how our research could be tailored to address the needs of the local management and community in West Hawai'i; challenges management faces when working with the communities; and their advice to our research project. We also attended local meetings to inform participants about the project and discuss relevant marine management issues. These meetings were intended to help build local support for future data collection and collaborative management.

Synthesis of Relevant Literature

We first searched the literature to understand how SES frameworks have been used to examine human well-being and cultural ecosystem services for natural resource and marine management. We focused on studies where social scientific methodologies might improve the representation of these concepts, especially related to measures of non-material elements and types of management interventions that might address them, as well as studies in Hawai'i or the Pacific Islands.

We then selected 11 key references most relevant to the West Hawai'i SES owing to their topical or geographical focus (Millennium Ecosystem Assessment, 2005; Smith and Clay, 2010;

Michalos et al., 2011; Dillard et al., 2013; Smith et al., 2013; Gould et al., 2014; Wongbusarakum et al., 2014; Biedenweg et al., 2016; Breslow et al., 2016; Pascua et al., 2017; Sterling et al., unpublished) and systematically reviewed them with respect to human well-being and cultural ecosystem services definitions and potential indicators. We created an initial list of the domains (broad category), attributes (definitions or descriptions of a category), and potential indicators of human well-being in relation to cultural ecosystem services used in each reference, grouping similar items together. The research team then discussed the terminology, underlying definitions, and examples of indicators in the references, further consolidating the groupings list. When there were differences, we relied heavily on the studies that had been conducted in Hawai'i or the Pacific Islands region (Gould et al., 2014; Pascua et al., 2017; Sterling et al., unpublished) and our own experience for their local relevance. We shared the results of our discussions with our mentors for feedback and discussion during group calls and through one-on-one discussions and revised accordingly.

Qualitative Data Collection

To learn from stakeholders and community members from West Hawai'i how to better incorporate human well-being aspects in coastal and marine management, we held an informal session at the Symposium on West Hawai'i's Marine Ecosystem in Kona, Hawai'i on December 6, 2017 and piloted a series of semi-structured informal interviews with community members.

The symposium was a free, 2-day event to which scientists, resource managers, and community members were invited to learn about ongoing research related to the regional marine environment. Our session was held over a 1½ h working lunch and was attended by approximately 25 individuals, primarily community members and resource managers. We began with a discussion of the ways in which human well-being is starting to be considered in ecosystem management and other IEAs, including the predominant depiction of human well-being as an outcome of ecosystem services. To begin to identify locally important connections between the marine ecosystem and human well-being, we then asked participants to reflect on the question "How does the marine ecosystem contribute to the things that matter most to the people in West Hawai'i?" We discussed this topic as a group and participants submitted specific written responses anonymously. The session revealed the importance of thinking about place-based conservation at a finer scale within West Hawai'i, described in the Section "Results."

Using discussion from the session as guidance, we focused our project on learning from communities that have organized around the ideas of place and conservation. We created a set of considerations to help identify candidate places and communities as project partners (**Table 1**). Rather than viewing these considerations as a checklist, we used them to reflect on benefits or challenges that potential communities might experience if they decided to work with us, as well as what our work would contribute to the communities. This process informed who we approached to be involved in the project and how we thought about desired outcomes.

TABLE 1 | Considerations for Identifying Place(s) and Community(ies) that can help ensure research process and outcomes have greatest benefit to all involved.

Cultural[1] conditions
1. Well-defined or clearly perceived boundaries of place by the community
2. Existing or reviving cultural and/or traditional practices or culturally valued locations
3. Support of community groups and agencies to continue the above practices and locations
Community conditions
1. High level of social cohesion and collaboration within the community
2. Diverse perspectives and opinions being well represented
3. Perception of community participation to other nearby communities
4. Level of transferability and useful lessons to others (researchers, managers, community members)
Research conditions
1. Possibility to build on existing relationships and strengthen trust between researchers and community
2. Availability of local champions (i.e., mentors or partners who work closely with the community and researchers)
3. Absence (or degree) of research/survey fatigue among community members
4. Availability of literature and secondary data from same or similar locations to use, learn from and replicate
5. Community *wants* this type of research, and can directly benefit from collaborating with researchers
6. Community *needs* this type of research for planning, community development, etc.
7. Existing foundational research and management in the place and with the community
8. Potential for future study
Governance/management conditions
1. Level of readiness of community members and leaders to work with resource management entities
2. Historical or on-going collaborative management efforts and successes
3. Management interest in conservation of particular place (e.g., aligns with management goals, ecologically or species-specifically significant)
4. Potential future management activities

[1]*Cultural conditions can refer to any community's traditions, practices, and important locations. They are not limited to Native Hawaiian culture but could include plantation culture, surfing culture, or fishing culture, for example.*

We also used the consolidated list that resulted from our synthesis of relevant literature as the basis for an interview guide. We reviewed in detail and pretested the interview guide with several project mentors, other researchers who conduct similar work with communities in Hawai'i, and community members. From April 23, 2018 to May 30, 2018 we conducted seven in-person semi-structured pilot interviews with leaders of communities in West Hawai'i that were working in conservation and place-based management. We asked about their relationship with their community(ies) within West Hawai'i, how they connect with the coastal and marine environment, their perception of the status of ecosystem, predictions they have for the future of their connections with the coastal and marine environment, and their thoughts on ways that science and marine management can help their community(ies) achieve its goals. Questions were open-ended and designed to gain a better understanding of the relevancy of each domain. Our consolidated list was used to prompt follow-up discussion, allowing us to

compare topics that they brought up themselves vs. following a rubric. Interviews lasted from approximately 1.25–2.50 h and were audio-taped.

Preliminary analysis of pilot interviews was conducted by one of the authors using NVivo 12 Plus (QSR International Pty Ltd.), primarily to check the relevance of the interview guide and domains and attributes used as prompts. When a larger number of interviews have been completed, we will complete a full analysis of interview transcriptions and notes.

Combined findings broaden our conceptual framework for thinking about the role of social dimensions in the IEA. Future work will continue interviews with a more diverse range of participants to gain a better perspective of how the coastal and marine ecosystem influences and contributes to human well-being, and to identify specific indicators of cultural ecosystem services and human well-being for West Hawai'i.

RESULTS

We present results in four main areas that improve frameworks to integrate the human dimensions of marine management into SES models for decision making: (1) insights on how to improve the representation of human dimensions within SES conceptual frameworks; (2) potential additional social indicators that might be included in West Hawai'i SES models; (3) the importance of place in relation to cultural ecosystem services and human well-being; and (4) depicting reciprocal and holistic aspects of SES models.

Representation of Human Dimensions in SES Conceptual Frameworks

We identified three areas where human dimensions, and in particular cultural ecosystem services and human well-being, were often underrepresented in the conceptual models used in coastal and marine and management: explicitly including the social system within analyses of the SES state; the interaction between biophysical and social conditions and ecosystem services; and the intentional use of socially oriented strategies to affect human behavior.

Many representations of SES in coastal and marine management use the term "ecosystem state" but measure only biological and physical ecosystem components. As previously mentioned, this approach assumes that with certain ecosystem states, an automatic flow of ecosystem services will result in human well-being. In this conceptualization, desired conditions (ecological health and human well-being) manifest at different points, where biophysical health is a relatively well-described and measured ecosystem state, and since human well-being depends on biophysical health, it is rarely measured as a separate outcome. IEA-focused models more clearly and intentionally ascribe human well-being at the same level of importance as ecological components (e.g., Levin et al., 2016). Thus, depending on the model, metrics of human well-being may be viewed as representing the state of the social system within an SES, or the state of the social system may be attributed to affecting human well-being outcomes.

When viewed through an ecosystem services lens, explicitly considering the state of the social system (e.g., food production and market structure, cultural norms, household characteristics, resource governance system, etc.) led us to think about the ways that social conditions can interact with biophysical conditions to access the benefits of ecosystem services, which are rarely discussed in the literature. For example, even with a service as straightforward as food provisioning, the presence of abundant fish stocks may be necessary, but not sufficient, to ensure food security. Social conditions, such as availability of fishing gear, food distribution networks, access to fishing grounds, and adaptive capacity of fishers may affect the extent to which fish are actually received as food throughout a community (Senapati and Gupta, 2017). In this conceptualization, the interaction between the state of the social system and ecological (biophysical) system determine the degree to which ecosystem services and related well-being are experienced. Given that many cultural ecosystem services are non-material and to a certain degree produced by society, we believe that the interaction between social and biophysical conditions/states will have especially important effects on cultural ecosystem services, and their equitable distribution, and should be examined in more detail.

Given the importance of social conditions on resultant ecosystem services, and therefore human well-being, we also noted that the representation of "ecosystem-based management" in models often did not explicitly discuss socially oriented strategies and outcomes, but rather focused on nature-oriented outcomes. In practice, managers often state, "We don't manage fish, we manage people," yet most models did not appear to have a clear way to represent management actions designed to affect the state of the social system that then cascade to effects on the biophysical system, although some ecosystem cascade models are including these reverse cascades (e.g., Spangenberg et al., 2014). Efforts such as campaigns to motivate participation in beach cleanups, inspire participation in voluntary data initiatives such as the saltwater angler registry, or adopt fishing practices that reduce harmful interactions with protected species instead are often presented broadly as education and outreach initiatives, even though the underlying intent may be to affect people's collective behavior. Explicitly identifying these initiatives as efforts to achieve a change in social conditions would bring attention to potential for social science disciplines such as social psychology or psychological anthropology to improve the effectiveness of these types of activities. These disciplines can improve understanding of the target audience's attitudes, knowledge, beliefs, and motivations to design and monitor campaigns that are more likely to result in the desired behavior, and related desired ecosystem results.

Human Well-Being Domains, Attributes, and Potential Indicators for West Hawai'i

Our review of relevant literature and discussions with mentors resulted in a consolidated list of human well-being domains, attributes, and potential indicators related to cultural ecosystem services tailored to West Hawai'i (**Table 2**). Here, domains represent the broad conceptual areas related to human

TABLE 2 | Human well-being domains, example attributes, and potential indicators for cultural ecosystem services in West Hawai'i.

Domains	Attributes	Potential indicators of cultural ecosystem services
Heritage	Multi-generational interactions/connections with natural resources Archeological and historic sites Cultural resources Acceptable historical change	Transmission of knowledge or practices around deified ancestral guardians (e.g., *'aumākua*); use or transmission of stories and verbal histories (e.g., *mo'olelo*); birth place and family burial sites; ceremonial practices, practices of respect, and other practices related to connection with place and resources
Spirituality	Interacting with the landscape to perpetuate spiritual beliefs and practices (e.g., divine power)	Formal ceremonial practices (e.g., *oli*, *pule*, other cultural protocols used to acknowledge relation to place); perpetuation of songs, chants, dances, and prayers of about place; protocols for place-specific gathering and harvesting practices
	Presence and recognition of plants, animals, and elements that represent/symbolize deities	Creation and use of ceremonial garlands (e.g., *lei*); ceremonial offerings such as fresh water, rain, salt, and turmeric
	Presence and recognition of familial guardians/ancestors; resources themselves recognized as kin	Recognition of deified ancestral guardians that are cared for by and take care of specific families (e.g., *'aumākua*)
Sense of place and identity	Sense of self, community, and/or home related to the coastal and marine environment	Activities on the landscape; heritage, social, and emotional connections to places
	Presence of historical place-based names which describe the past and present of the coastal and marine environment	Place names; landscape terms; species names; environmental process names (e.g., rain names, wind names); transmission of existing or creation of new cultural proverbs to describe these observations
	Engagement of families in coastal and marine resource based activities	Existence and availability of activities such as fishing or harvesting for livelihood or enjoyment
	Presence on and interaction with lands that will remain secure (formally or informally) for future generations	Presence by lease, physical access, ownership, and/or occupation; customary rights and tenure
Education	Local knowledge about the coastal and marine environment	Language and/or culture encoded knowledge of seasonal patterns such as timing and intensity of rain and other meteorological phenomena or plant/animal behavior and reproductive cycles; place-specific practices associated with storied landscapes
	Knowledge transmission (place-based, observational, formal, informal, etc.)	Scientific research, experiential, land-based education, learning from elders, culture-based education (e.g., gathering salt from natural pools and making salt in raised ponds)
	Presence of environmental signs or indicators (e.g., bioindicators) and the ability to recognize them	Species or environmental processes that signal the cycles of another plant/animal species (e.g., types of rainbows to signal events)
Social relations	Perpetuation of practices/skills that allow individuals to provide for and share with their families and community	Goods for household, sharing, and income; jobs that require knowledge of traditional practices or the discipline required; formal and informal apprenticeships; place-based fishing/gathering practices; community fishing endeavors; acknowledgment of young leaders
	Presence of strong social ties or networks; sense of community; trust in neighbors	Network of people to share with and receive from; gifting/exchanging of goods; joint family endeavors; communal child care; community spaces
Stewardship	Ability to care for resources and environment	Contributions of time, labor, and/or monetary support toward maintenance of public or private lands or specific sites; restoration and maintenance of sacred sites (e.g., *wahi pana*), civic activities around public spaces
	Customary rights and responsibilities are locally known, practiced, and respected	Recognition and use of access restrictions, gathering rights, and easements related to traditional ownership or harvesting practices (e.g., *kapu*)
Existence	Aesthetics	Recognition and practices around the appropriate maintenance of specific sacred sites; pride in community parks and coastal areas; beach clean up activities
	Inspiration	Broadly circulating public discourse about collective responsibilities (e.g., caring for place or *malama 'aina*)
	Creativity	Local artistic or creative practices; moralization; poster competitions in schools
Governance and management	Political participation and equity	Participation in marine management decision-making processes and leadership; stakeholder processes; exercising rights/interest in politics; management reflects local and traditional values
	Effectiveness of management	Perceptions of management, permits, and regulation; adequate funding and staff capacity for achieving management objectives; partners and collaboration
Health	Physical and nutritional health Mental and emotional health	Outdoor activities that promote health and strength of body and mind
Safety and security	Security and safety related to real or perceived environmental risks	Protection from threats of natural disasters such as hurricanes, tsunamis, earthquakes, etc. (e.g., level of social preparedness for natural disasters; access to social nets; availability and application of traditional knowledge to mitigate environmental risks)

well-being. Each domain has associated attributes that further describe its aspects and characteristics. Potential indicators are variables associated with the attributes. Some indicators are effective for multiple associated attributes. Later, metrics can be developed to measure the state of selected indicators to assess changes or evaluate trade-offs related to potential management actions. Many Native Hawaiian examples are included because the cultural ecosystem services work in this region has focused on Native Hawaiian culture (see Gould et al., 2014; Pascua et al., 2017); however, most of the domains and attributes can be applied to any of the communities in West Hawai'i described in the introduction.

The definitions of cultural ecosystem services and human well-being emphasize the relationships and meanings derived from interactions with the environment. As described in Satterfield et al. (2013), studies often emphasize the tangible, or material, aspects of the interactions as suggested indicators. These may or may not reflect relationships that are not generally perceived as tangible, but which nevertheless play materially impactful (causal) roles in environmental and ecological states and outcomes. We modified items in our consolidated list to reflect this relational aspect. For example, in the 11 key references we synthesized, recreation is typically listed as a cultural ecosystem service and component of human well-being, with potential metrics including number of visitors, related-jobs, and income generated through recreational activities. Yet, we saw recreation reflected in multiple well-being domains, such as physical health related to outdoor activities, mental and psychological health from regaining life balance and shared activities with families and friends, or landscape-based activities that are related to sense of place. We viewed the aspects of the activity related to cultural benefits, rather than the activity itself *per se*, as a better gauge of the cultural ecosystem service it provided. As we move to the full data collecting stage of the project (completing interviews with community members), we will listen for specific examples that will help us better represent the meanings related to the biophysical components of the landscape and activities in which people engage. For example, rather than merely counting kayak trips, we will also inquire into motivations and experiential benefits of kayaking to those engaged in the activity.

This focus on meanings, relationships, and importance of activities also underscores the usefulness of a bigger toolkit drawn from many social science disciplines to identify appropriate metrics for non-material contributions of ecosystems to human well-being. Primary data collection would be necessary for crucial indicators of the cultural ecosystem service aspects of well-being, such as range of emotional connections to places, amount of pride in community parks, or perceived degrees of protection from environmental risks. While these concepts refer to experiential phenomena, psychometric scales can be created to systematically evaluate the degree to which populations experience them. Additionally, place-based stories, ethnographic narrative, and qualitative analyses can provide in-depth understanding of the meaning of well-being and relationships between social and ecological systems (Vaughan, 2018).

Importance of Place in Relation to Cultural Ecosystem Services and Human Well-Being

As previously mentioned, participants in our informal session at the Symposium on West Hawai'i's Marine Ecosystem emphasized the importance of investigating place at a finer spatial scale. They were uncomfortable treating all of West Hawai'i as one community, noting that specific geographies within West Hawai'i will lead to different types of interactions between communities and marine resources. For example, the extent of coral cover or the influence of submarine groundwater on coral reefs near a community's shoreline result in different ecological characteristics that are conducive to different types of activities and resultant meanings. Attending to place was not only a large part of the session dialogue, it was also reflected in a word cloud created from the written responses to the discussion question (**Figure 2**). These observations led us to focus our research on better understanding reciprocal relationships in a place-based conservation context. The importance of place was also evident throughout the pilot interviews. As suggested in our consolidated list of potential indicators, characteristics of place were reflected in the way people talked about potential indicators. This highlights the importance of not only taking a place-based approach to ensure relevancy of results, but also the way that the geography and ecology of the place itself factor into specific elements of well-being.

There was an assumption by many that because the West Hawai'i IEA is at a smaller scale than other IEAs (e.g., only one part of one state), it would be simpler to identify indicators of social phenomena such as human well-being. Yet, although relatively small in spatial scope when compared to other IEA regions in the United States, West Hawai'i is comprised of multiple unique places which may require site-specific indicators. The West Hawai'i IEA is improved by working closely with these unique communities to identify site-specific management needs. We observed that analogous to the way coastlines exhibit fractal characteristics, with similar spatial patterns revealed at different scales, stakeholder engagement exhibits similar fractal qualities. That is, stakeholder engagement to identify social indicators is equally complex at multi-state levels, vs. local place-based levels. However, the composition of stakeholders will change based on the management questions, which also vary by scale.

Reciprocal and Holistic Social-Ecological Systems Models

In addition to identifying the importance of place, participants in the symposium session also expressed concern with conceptualizing human well-being as an outcome of ecosystem services, as is often depicted in SES models. They explained that viewing human well-being in this way does not adequately convey the reciprocal connections between people and the land and ocean. As one participant described, "...if place is healthy we are healthy. We make place – place makes us. It is in us – our food, livelihood, identity, purpose in life." Reciprocity was also evident throughout the interviews as the natural way that people



FIGURE 2 | Word cloud of compiled responses from all participant responses to the question "How does the marine ecosystem contribute to the things that matter most to the people in West Hawai'i?" All responses included are verbatim. Word size relates to frequency of word use.

talked about their relationships with the environment and/or how they connect with the environment.

Discussions about reciprocal relationships between people and the marine ecosystem led us to think critically about the linear or cascading models of ecosystem services that portray people primarily as negative stressors. Some SES models add people as beneficiaries of positive ecosystem goods and services to represent reciprocity (e.g., Kittinger et al., 2012; **Figure 3A**), however, this relationship only represents one dimension of reciprocity. Similar to other studies (e.g., Rissman and Gillon, 2017; Vaughan et al., 2017; Winter et al., 2018), participants in our research instead described a system where people could be both environmental stressors and stewards, and experience both benefits and risks from the ecosystems (**Figure 3B**). Examples of ecosystem risks in West Hawai'i include king tides, storm surges, vog (air pollution from volcanic gasses), and lava flows.

In addition, our work illuminated a tension between scientific models that parse out elements of the system and the more multivariate dynamics of human domains within the system. Interviewees rarely described a single human well-being domain when discussing interactions with the ecosystem, even when prompted with a question designed to relate to a single domain or attribute. While modelers tend to refer to considerations of a holistic system in terms of identifying all the discrete elements within the system, interviewees described a more gestalt experience where the elements were experienced as broadly interactive, as in **Figure 3B**, making them difficult to separate.

DISCUSSION

As natural resource managers increasingly move toward ecosystem-based approaches and SES frameworks, metrics of human well-being and cultural ecosystem services will be necessary to determine success of management interventions. Yet, there have been considerable challenges in including and

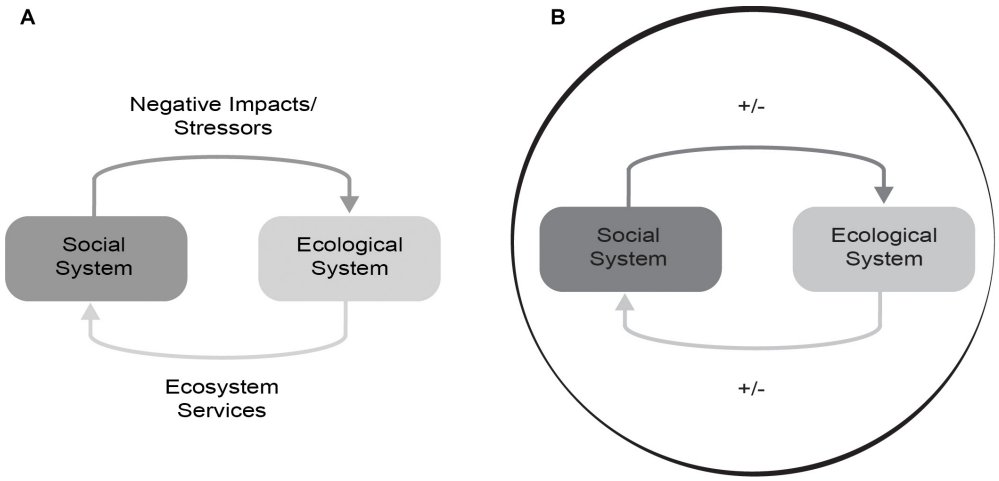


FIGURE 3 | (A) One-dimensional depiction of reciprocity within a social-ecological system, with the social system exerting pressure on the ecological system (biophysical components) and receiving benefits through ecosystem services (based on Kittinger et al., 2012). **(B)** Emergent stakeholder perception of reciprocity, where the social system and ecological system experience both positive and negative interactions with each other, and people generally experience the system as a whole.

operationalizing these concepts in SES models for coastal and marine management. This study contributes key insights and questions focused on: (1) points of inclusion for human dimensions in SES models, (2) culturally relevant domains of human well-being and related indicators, (3) the importance of place and its interaction with scale, and finally (4) the tension between a gestalt vs. discrete approach to modeling, assessing, and sustainably managing SES.

Our examination of SES frameworks identified several points where attention to human dimensions are typically under-represented. First, it is unclear when and how human well-being should be considered a social system state. On the one hand, human well-being may be considered the desired outcome of a management action, and therefore representative of the state of the social system. On the other hand, the state of the social system may be seen as interacting with the state of the biophysical system in delivering ecosystem services that affect human well-being. The conceptualization chosen has implications for identifying and monitoring indicators, as well as planning and implementing management interventions. Spangenberg et al. (2014) and Schleyer et al. (2017) discuss ecosystem services as anthropogenically defined and produced, where the actual benefits received depend on the social and ecological interactions. It may be important to consider the reflexive influence of human well-being conditions (as one aspect of the state of the social system), or to view ecosystem cascade models as representing multiple time steps related to state of the social system. Assuming that the presence of biophysical conditions will result in ecosystem services without considering these social interactions not only misses opportunities for potential social interventions, it also ignores the social processes that, in many situations, may be necessary to ensure fair and equitable distribution of these services. Including these interactions in SES conceptual models may help identify other socially directed management strategies necessary to ensure sustainable and equitable receipt of ecosystem services, and related human well-being outcomes.

In addition, SES frameworks would benefit from more clearly including socially oriented strategies and outcomes. For much of natural resource management, including marine management, socially directed management strategies are often not explicitly designed to target behavioral change and positively affect biophysical conditions, but are instead limited to education and outreach to build awareness. Lack of exposure to social science disciplines may cause managers to overlook other promising and creative approaches to encourage conservation behaviors. For example, the discipline of conservation marketing is now being recognized as a key area of social science contribution to conservation practice (Bennett et al., 2017). This discipline applies conservation psychology and traditional marketing techniques to increase participation in pro-environmental behaviors and reduce activities that negatively affect the environment. Explicitly recognizing when social interventions are intended to change behavior, and understanding their socio-cultural values and relationships with the natural resources, can help identify additional resources, strategies and partnerships that may result in more effective management.

In addition to broad SES frameworks, our consolidated list of human well-being domains, attributes, and potential indicators can help managers identify areas requiring actions to improve elements of well-being related to cultural ecosystem services. To effectively measure the effects on non-material aspects of these concepts, primary data may need to be collected, using social science methodologies. Researchers outline a number of techniques which draw from the full range of social science disciplines and practice areas, including topics as diverse as ethnography, economic

valuation, deliberative governance, and participatory mapping (Daniel et al., 2012; Fish et al., 2016; Small et al., 2017). This range indicates the importance of including a diversity of social scientists from multiple backgrounds as part of an IEA team.

When we started this project, there was an assumption that the smaller scale of the West Hawai'i IEA, relative to other NMFS IEAs, would simplify stakeholder engagement. Yet, we observed that engagement with communities revealed fractal-like characteristics. Large ecosystem scale science and management (e.g., at state or large marine ecosystem levels) is understood as complex, yet smaller geographies (e.g., local and place-based systems) are no less complex, socially and ecologically. Instead, different management questions may be relevant. Small et al. (2017) describe how levels of social organization, from individual, to groups, to communities, to society, affect the values attributed to ecosystem services. They also identify different socioeconomic and environmental drivers of change that act at different spatial and temporal scales and recommend taking a multi-scalar approach. Following such an approach may help identify the types of management questions that can be reasonably addressed at different scales, as well as the range of stakeholders who can be practically engaged. Insights from the local place-based scale of research we explored in this project are necessary to inform management at the site level, but may require additional science investments to carry out across a larger-scale IEA. To make comparisons across sites possible, we recommend following the advice of Breslow et al. (2016), in selecting a set of core metrics alongside site-specific metrics. The core metrics allow transferability and comparison across regions, while site-specific metrics ensure relevant place-based indicators are present at the management table.

Finally, our project is ultimately focused on eliciting specific social metrics of human well-being and cultural ecosystem services, which we believe must be included in scientific models if we are to more effectively and comprehensively assess SES. However, we also identified discrepancies in the way these models tend to portray the relationship between the social and ecological components of the system and the way they are experienced by community members. First, many models portray a one-dimensional view of reciprocity between human communities and marine ecosystems, which focus on benefits to people and may miss important considerations of vulnerability (Binder et al., 2013). In addition, people living in these systems do not experience them piecemeal, but rather as a whole. Viewing/experiencing the environment through this reciprocal, holistic lens is not unique to traditional and indigenous societies, although it been most well studied in these contexts (c.f., traditional ecological knowledge research, Berkes et al., 2000). However, reciprocal and holistic concepts are rarely integrated into modern resource management despite being central cultural models among many contemporary communities. This mismatch is particularly challenging for place-based management, where managers want and value local community input, but management tools may not seem relevant to community experiences and needs. Yet, without uncovering all components of the system, crucial considerations related to cultural ecosystem services and human well-being run the risk of being overlooked by management. Guidance from the fields of community-based management, co-management, and stakeholder engagement (for example, see Wondolleck and Yaffee, 2000, 2017; NOAA Office for Coastal Management, 2015; Hawai'i Sea Grant, 2018) may be useful in bridging the needs of scientific modelers for discrete and simple components and the holistic, boundary-collapsing gestalt of living in a place.

There were a number of limitations to this first stage of our research. First, while we focused on frameworks typically used in marine and natural resource management, we recognize that there is a large body of research around human well-being in other contexts from economic growth to international development (for review, see Dodge et al., 2012; Biedenweg et al., 2016). Our work focused specifically on the linkages between human well-being and coastal and marine resource management. Given that we identified limitations in how these concepts have been applied in practice, it would be useful to re-examine some of the questions raised by our study through the lens of well-being used in other contexts, including drawing from disciplines such as public health, psychology, and economics as we refine our research. For example, recreation as a domain of well-being may be unique to natural resource management, whereas our interpretation of recreation as an activity that influences well-being may be more similar to broader literature under which recreation may impact human well-being domains such as physical and mental health. Second, our focus on the linkages between cultural ecosystem services and well-being does not include aspects of well-being related to the other main categories of ecosystem services, which need to be included to fully assess well-being in a SES. Many of those linkages are already accounted for in current conceptual models of the West Hawai'i ecosystem (Ingram et al., 2018), however, it is possible that examining cultural ecosystem services separately from the other ecosystem services may affect the relative importance placed on different types of services. Finally, we did not yet assess the status of potential indicators of well-being to determine those most sensitive to management actions. This will be a necessary future step to identify a set of meaningful and manageable set of metrics to monitor.

Despite these limitations, our work enhances recent efforts to improve the representation of the human dimensions of SES. By advancing our thinking about the broad frameworks used to represent cultural ecosystem services and human well-being into SES models, we are improving the ability to achieve NMFS guiding principles related to ecosystem-based management, especially related to appropriate social indicators. Future work will apply these insights with partner communities to identify more specific indicators. We hope that our findings not only improve the ability of future models to assess status and trends of the full range of SES components, but also to holistically integrate human experiences into the management of marine ecosystems, large and small.

ETHICS STATEMENT

This study was carried out in accordance with the recommendations of the University of Hawai'i Institutional Review Board (IRB) with written informed consent from all subjects. All subjects gave written informed consent in accordance with the Declaration of Helsinki. The project has exempt status for Human Subjects Research from the University of Hawai'i Committee on Human Studies (CHS) under the exempt project 19449, Socioeconomics of Western Pacific Fisheries.

AUTHOR CONTRIBUTIONS

KL and SW conceived the project, wrote the funding proposals, and developed and coordinated the project design and activities. RI was responsible for day-to-day project management, contributed significantly to project design, and conducted the literature review and interviews. KL and RI co-led the symposium session. AM and MP served as project mentors and helped shape the direction of the project. KL led the writing of the manuscript. All authors contributed significantly to the writing of the manuscript.

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How Does the Diversity of Divers Affect the Design of Citizen Science Projects?

*María I. Hermoso[1,2], Victoria Y. Martin[3], Wolfgang Stotz[1], Stefan Gelcich[4] and Martin Thiel[1,2,5]**

[1] Facultad Ciencias del Mar, Universidad Católica del Norte, Coquimbo, Chile, [2] Millennium Nucleus Ecology and Sustainable Management of Oceanic Island (ESMOI), Coquimbo, Chile, [3] Cornell Lab of Ornithology, Ithaca, NY, United States, [4] Center of Applied Ecology and Sustainability, Pontificia Universidad Católica de Chile, Santiago, Chile, [5] Centro de Estudios Avanzados en Zonas Áridas, Coquimbo, Chile

Divers have widely participated in citizen science (CS) projects and are one of the main groups of marine citizen scientists. However, there is little knowledge about profiles of, and incentives for potential divers to join CS projects. To date, most studies have focused on the SCUBA diving industry; nevertheless, there is a diversity of divers, not all using SCUBA, who engage in different activities during their dives. Differences in diver profiles could affect their willingness and ability to contribute to CS. In this study, we compare the diving profile, interests, preferences and motivations to participate in CS of five diver types (artisanal fishermen, recreational divers, instructors, scientific divers, and others). All divers have strong interests in participating in CS projects, with no major differences among diver types. In general, they are interested in a wide variety of themes related to CS but they prefer simple sampling protocols. Divers are motivated to participate in CS to learn about the sea and contribute to science. Some important differences among diver types were found, with artisanal fishermen having significantly more dive experience than other diver types, but less free time during their dives and limited access to some communication channels and technologies. These characteristics make them ideal partners to contribute their local ecological knowledge (LEK) to local CS projects. In contrast, recreational divers have the least experience but most free time during their dives and good access to cameras and communications channels, making them suitable partners for large-scale CS projects that do not require a high level of species knowledge. Instructors and scientific divers are well-placed to coordinate and supervise CS activities. The results confirm that divers are not all alike and specific considerations have to be taken into account to improve the contribution of each diver type to CS. The findings provide essential information for the design of different types of CS projects. By considering the relevant incentives and opportunities for diverse diver groups, marine CS projects will make efficient gains in volunteer recruitment, retention, and collaborative generation of knowledge about the marine environment.

Keywords: participatory science, subtidal, SCUBA, fishermen, recreational divers

Abbreviations: CS, citizen science; LEK, local ecological knowledge.

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***Correspondence:**
Martin Thiel
thiel@ucn.cl

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INTRODUCTION

Importance of Public Preferences for Participation in Citizen Science

Citizen science (CS) is a way to generate new scientific or environmental knowledge, through the involvement of members of the public in scientific research (Dickinson et al., 2012). A variety of studies have demonstrated the usefulness of CS projects for scientific research and have also shown that CS provides other socio-ecological outcomes (Shirk et al., 2012; McKinley et al., 2017). For example, participating in CS projects enhances people's knowledge about wildlife (Brossard et al., 2005) or about science (Bonney et al., 2015), and even induces changes in environmental management (Danielsen et al., 2005, 2010). Also, changes in people's attitudes toward science or the environment have been demonstrated (Price and Lee, 2013), but are not widely investigated (Brossard et al., 2005). One socio-ecological outcome considered important in conservation practice is to engage communities in actions to mitigate environmental problems (McKinley et al., 2015). Contributing to CS efforts presents one type of action people can take. However, evidence suggests the people most interested in CS programs are those already concerned about science and the environment (Martin, 2017), meaning that CS will naturally attract a particular type of participant (pro-science or pro-environmental) rather than a broad reach across audiences with many different science/environmental attitudes. Furthermore, one of the challenges in CS projects is to recruit and retain participants. For that purpose, it is important to understand potential volunteers (Cigliano et al., 2015; Aristeidou et al., 2017) and their needs and interests in CS. In general, there is only limited knowledge about the preferences of non-participants (Martin et al., 2016b), which is nevertheless essential information to achieve a broader and more diverse participation of citizen volunteers in CS projects.

Divers are one of the main volunteer groups participating in marine CS projects (Thiel et al., 2014) and one of the most interested marine user groups (Martin et al., 2016b). However, little knowledge exists about the profiles of divers who join CS projects (Cerrano et al., 2016) and even less information is available about those who are not joining. Here we present a study of different types of divers in Chile, their background characteristics, diving experience level and their preferences for different ways of participating in marine CS.

Lack of Information About the Diversity of Divers in Citizen Science Projects

Existing evidence suggests that divers participating in CS projects have a high education level, previous interest in science and high diving experience (Arvanitidis et al., 2011; Cerrano et al., 2016; Lucrezi et al., 2018). Martin et al. (2016b) studied public interest in marine CS and found that SCUBA divers have the highest level of interest in CS among many marine user types. A recent study by Lucrezi et al. (2018) found a generally high interest among SCUBA divers to participate in CS, although only a few actively participate. Furthermore, they found a greater proportion of divers with professional certifications had participated in CS than basic divers. Although previous studies generated important knowledge for CS design (e.g., Cerrano et al., 2016; Martin et al., 2016b; Lucrezi et al., 2018), they focused primarily on the recreational SCUBA diving industry, excluding other groups who either work or recreate underwater, such as fishermen (especially those using hookah) and free divers (snorkelers). These groups possess the potential to contribute to the same CS projects that aim to engage SCUBA divers.

The diversity of divers can be differentiated by their diving technique (SCUBA divers, snorkelers, hookah, etc.) or based on their main underwater activity or profession (e.g., fishermen, military, commercial). For example, recreational divers dive for fun, adventure and to enjoy marine life (Musa et al., 2006; Edney, 2012), and may have free time for CS projects during their dives. In contrast, dive instructors dive primarily to teach or guide other divers. Given the responsibility they have, their time is limited to make and record observations during their regular dives. Some divers also extract marine resources, either professionally as artisanal fishermen (Godoy et al., 2016) or for their own consumption (Young et al., 2015). The main purpose (recreational or professional) and activities undertaken during dives is expected to strongly influence whether and how different types of divers might participate in CS.

The diver types also differ in other aspects such as demographics, experience, or interests. For example, the majority of recreational SCUBA divers are males, well educated, have high incomes, and around 5 years of diving experience (Edney, 2012; Lucrezi et al., 2013). Dive instructors usually have advanced skills but are of a similar age as recreational SCUBA divers (Watson and Pulford, 2004). Tessier et al. (2015) found recreational divers have a lower age and less experience compared to spear fishermen and other marine users. Martin et al. (2016b) found differences among stakeholder groups in their preferences to participate in CS, and while all groups in that study listed data collection as their most preferred CS activity, the groups differed in their second preference. After data collection, fishermen preferred to assist formulating research questions, divers preferred participating in data analysis, and other groups preferred to communicate the findings. Thus, it is important to take the personal and professional history of divers into account when creating CS projects to involve groups in different ways.

Diving experience can also lead to differences in divers' perception, knowledge, and attitudes about the marine environment and management (Todd et al., 2000; Thapa et al., 2005; Dearden et al., 2006; Salim et al., 2013). Recent evidence suggests that opinions or preferences about marine management issues may differ among marine users such as fishermen, recreational anglers, divers, etc. (Hattam et al., 2014; Wiener et al., 2016). Background characteristics (e.g., attitudes, knowledge, and values) held by different diver types, in combination with the demands of their diverse underwater activities, are likely to affect their willingness and ability to participate in CS. Therefore, it is important to know more about

the differences among diver types in order to gain a better understanding of their potential and requirements to participate in CS. This is key to inform project design aimed at increasing diver participation and diversity in CS.

Citizen Science Project Requirements for Divers

Some CS projects ask for a specific profile of diver according to the tasks required to be undertaken. For example, Reef Life Survey (RLS)[1] or Reef Check[2] ask for recreational SCUBA divers, as their protocol requires autonomy (e.g., protocols required to be performed during the dive). Other more flexible projects are open to most marine users, such as Redmap[3] or Reef Environmental Education Foundation (REEF)[4], which use less complex methodologies. Other projects require specific gear, for example, a dive computer to report seawater temperature (Wright et al., 2016) or a camera to take photographs that are then submitted to the CS platform, e.g., iNaturalist (Jacobs and Zipf, 2017). Some projects also require a minimum level of diving experience, for example, Reef Check or RLS (Edgar and Stuart-Smith, 2014; Roelfsema et al., 2016) or a minimum level of knowledge (verified by an exam) before volunteers can join (e.g., REEF; Wolfe and Pattengill-Semmens, 2013).

The type of CS project also influences the type of divers who participate. For example, fishers (recreational or professional) participate in initiatives about fisheries resources (Godoy et al., 2010; Suazo et al., 2013; Coll et al., 2014; Thiel et al., 2014), while SCUBA divers (recreational or instructors) participate in studies of a wider variety of target species (Schmitt et al., 1993; Edgar and Stuart-Smith, 2014) or focus on charismatic species (Theberge and Dearden, 2006).

Protocols used by recreational divers are normally based on direct observations of species during surveys, transects or opportunistic observations, while studies with fishers are frequently based on interviews or questionnaires (Thiel et al., 2014). SCUBA divers are often trained to apply their methodology before participation, while specific training sessions are rare in studies with fishermen (Schmitt et al., 1993; Arvanitidis et al., 2011). Yet these different approaches to volunteer engagement are not usually based on an understanding of the preferences different diver types have for their involvement in marine research.

Since it is clear that diver types differ in many aspects, it is reasonable to suggest they will also differ in their preferences for participation in CS. This study aims to investigate these differences to improve the design of marine CS projects with divers. Specifically, our research question is: *Does diver type affect the ways in which different groups of divers might engage with CS*? To answer this question we characterize different diver types to determine their needs, capacity, interest and potential to participate in CS projects, focusing specifically on their diving experience, preferences and motivations.

[1] www.reeflifesurvey.com

[2] www.reefcheck.org

[3] www.redmap.org.au

[4] www.reef.org

Case Study Area

Chile represents an opportunity to test the research question, as it is a country with an extended coastline and a wide variety of divers with different backgrounds who could potentially participate in CS. Despite the diversity and large number of people diving along the Chilean coast (Godoy et al., 2010; Aburto et al., 2013; Biggs et al., 2016), no previous study has used a comparative approach to determine the potential of the different diver types to engage in CS projects.

There are at least five diver types in Chile (e.g., Godoy et al., 2010, 2016; Biggs et al., 2016): (i) artisanal fishermen, who dive to collect seafood for the market and typically dive with the air supplied directly through a hose from a compressor on the boat (hookah) (Godoy et al., 2016), (ii) spear fishermen, who despite diving without air support (i.e., free dive) spend extensive time underwater and dive to great depths to capture fish recreationally (Godoy et al., 2010) or professionally (Godoy et al., 2016), (iii) recreational SCUBA divers who dive for fun (Biggs et al., 2016), (iv) dive instructors who guide or train other divers, and (v) scientific divers, who have a scientific background or who dive primarily for research. Furthermore, there are commercial or military divers, who are not our target group in this study as their diving activity precludes their potential participation in CS.

In this study, we examine the hypothesis that "divers" are a diverse group whose differences in experiences, interests, and preferences to participate in CS will vary. In order to test this hypothesis, we surveyed a diversity of divers along the Chilean coast to describe different diver types and their preferences regarding participation in CS.

MATERIALS AND METHODS

Sampling

A series of structured face-to-face interviews (Bryman, 2012) were conducted in fishermen's coves, diving centers, and universities from Arica to Chiloe between January and March in 2017 (**Table 1**). A convenience sampling of divers was carried out at the different sites (fishermen's coves or diving centers) where many divers were asked whether they would be willing to participate in an interview. Due to the low numbers of divers in some locations, it was necessary to use convenience sampling instead of a random sampling method. In each diving center, at least one dive master or instructor was interviewed. To find scientific divers, a "snowball sampling" method was used (Bryman, 2012), whereby scientific divers at universities (mainly in marine science faculties) were asked to help identify other scientists who dive. Before starting the interview, the interviewees were told the main goal of the study was to characterize divers in Chile and evaluate their interest in participating in CS projects. When inviting people to participate in the interview, we also highlighted they did not have to answer questions they felt uncomfortable with. In total, 229 divers were interviewed and only three people declined the interview, which represents a response rate of 99%. All the people who declined were artisanal fishermen who were busy working. All interviewees in this study formally agreed to participate in the study.

TABLE 1 | Number of divers surveyed in each location.

Location	N	Location	N
Achao	2	Los Molinos	2
Algarrobo	6	Los Molles	15
Ancud	5	Los Vilos	1
Antofagasta	18	Mejillones	2
Arica	21	Niebla	7
Bahía Inglesa	7	Osorno	4
Bahía Mansa	1	Pichidangui	8
Caldera	8	Pisagua	8
Carelmapu	3	Playa Blanca	1
Chañaral de Aceituno	7	Pucatrihue	2
Concepción	14	Puerto Montt	5
Copiapó	1	Puerto Varas	5
Coquimbo	13	Qellón	4
Corral	1	Quintay	11
Dichato	5	Tomé	3
Horcon	2	Totoralillo (Coquimbo)	1
Iquique	12	Valdivia	6
Las Cruces	6	Valparaíso	5
Lirquen	3	NA	2

Survey Instrument

The questions reported here were part of a larger survey of divers in Chile. The specific interview questions described in this study are presented in the **Supplementary Data Sheet S1**. The questions were designed to understand the diversity of divers assessing diving profiles, preferences and motivations for participation in a CS project. The questions were divided into five main sections: (a) demographics and contact, (b) diving profile, (c) general interest and preferences in CS, (d) protocol preferences, and (e) motivations to participate in CS.

Demographic questions (a) were developed to determine differences in age, nationality, education and gender to understand other differences within groups. For diving profile characterization (b) we asked about the diving technique used (apnea, SCUBA tank, hookah, or other), diving experience indicator (total of years diving), and about their free time during a dive. In addition, divers were asked whether they own and/or use particular accessory equipment (underwater camera and dive computer). To study general interests and preferences in CS (c) questions were subdivided in different items (general interest in participating, preferences in particular science fields, in developing specific scientific skills, studying marine biodiversity, and preferences in training aspects such as "willingness to pay for training" or "having to pass an exam"). Responses to questions in this section were measured on 5-point scales. For preferences in the protocol (d), the respondents were shown pictures of six typical protocols used in CS projects to ensure they understood what was meant by each type (**Figure 1**). The protocol types were scaled from less demanding (opportunistic observation) to more demanding (conduct underwater transects or experiments). To measure protocol preferences a ranking system was used to score each type of protocol. Interviewees were asked to rank these options from 1 (most preferred) to 6 (least preferred); it was possible to score 0 for protocols that the diver would never want to do. For data analysis we made a conversion from 6 (most important) to 1 (least important), and left the 0 for those protocols that divers did not want to do.

For motivation to participate in CS projects (e), respondents were asked to rank the importance of five different motivations. The motivational choices were based on the most cited motivations for volunteers (Bruyere and Rappe, 2007; Measham and Barnett, 2008; West and Pateman, 2016). Options of responses were "Learning about topics related to the sea," "Meeting people with my own interests," "Feeling that I contribute to the study of the seabed", "Having fun in the activity," and "Obtaining discounts on diving equipment"; interviewees had to rank these options from 1 (most important) to 5 (least important). Similar as above for protocols, here we inverted the scale for data analysis, so that the highest value (5) means "most important" while the lowest value (1) means "least important."

Classification of Diver Types

In the section on diver characterization (b) interviewees were asked to identify the one diver type they most identify with. The options were: artisanal fishermen, recreational divers, instructors (dive masters are included), scientific divers, spear fishermen, commercial divers, and "various" (described below). Of the 229 divers interviewed, 72 characterize themselves as artisanal fishermen, 48 as an instructor, 66 as recreational, 22 as scientific, 15 as spear fishermen, and 6 as commercial. Since the number of commercial divers and spear fishermen is low, these categories were collapsed into the group "various." Finally, there were some difficulties in the classification of scientific divers. Scientific diver is a complex concept as there are some divers who have a specific certification for scientific diving but have no professional scientific background or training (ManeyJr., and Genovese, 2000), while others are scientists but do not dive for research purposes. In this study, we defined the scientific divers as not only divers for whom their main activity during a dive is to do scientific work, but also include divers who have a scientific background. This means all divers who studied natural science at the university level are herein considered as scientific divers, irrespective of whether they classify themselves as such. The final categories evaluated include a total of 71 artisanal fishermen, 34 dive instructors, 52 recreational divers, 58 scientific divers, and 14 "various."

Data Analysis

In the present study, we used descriptive statistics to characterize divers, then tested for differences among different diver types in particular aspects (diving profile, interest, preferences, and motivation to participate in CS). Statistical analyses were undertaken with R studio (RStudio Team, 2016). Normal distribution of variables was tested using both the Shapiro–Wilk and Kolmogorov–Smirnov tests for normality. As most variables were not normally distributed, non-parametric analyses were used (Field, 2013). To detect differences among diver types Kruskal–Wallis and Mann–Whitney *post hoc* tests (Dytham, 2003) were performed. Effect sizes for each significant difference were calculated using Cohen's d

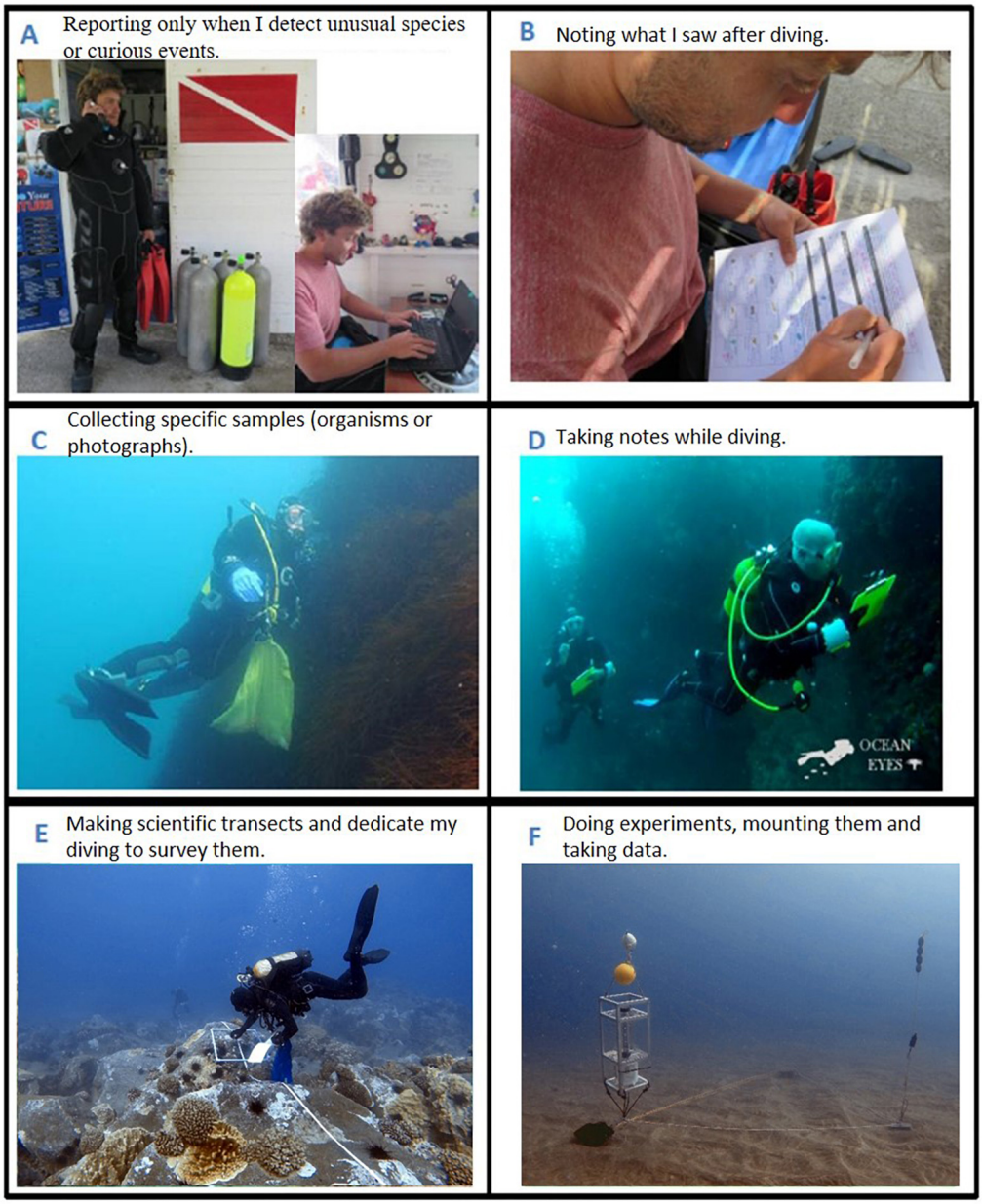


FIGURE 1 | Pictures of the six different protocols presented to divers. A written informed consent was obtained from the individual for the publication of this image. **(A)** Reporting only when I detect unusual species or curious events. **(B)** Noting what I saw after diving. **(C)** Collecting specific samples (organisms or photographs). **(D)** Taking notes while diving. **(E)** Making scientific transects and dedicate my diving to survey them. **(F)** Doing experiments, mounting them, and taking data.

(see **Supplementary Table S1**). A principle component analysis (PCA) was conducted to identify main variables that differentiate the diver types.

Limitations of the Study

Even though in this study great effort was made to distinguish the main types of divers in Chile, it would still be possible to further segregate some groups of divers or to subdivide them in other groups. For example, scientific divers could be subdivided into those who have a scientific background and those who are employed specifically to do scientific sampling during their dives (whether or not they have a scientific background). On the other hand, there are photographers who, in this study, were distributed mostly among the recreational and scientific divers, but could also be placed in a separate group of submarine photographers. Furthermore, spear fishermen are underrepresented in this study, but make up most of the "various" category. Although we attempted to interview similar numbers of divers for each group, some diver types were easier to identify and locate than others. For example, artisanal fishermen can

TABLE 2 | Demographic characteristics of diver types.

	Artisanal fishermen *N* = 71	Dive instructors *N* = 34	Scientific divers *N* = 58	Recreational divers *N* = 52	Various *N* = 14
Nationality (%) Chilean	100%	94%	93%	90%	100%
Gender (%) men	87%	76%	78%	67%	100%
Age (Average ± SD)	48 ± 9	34 ± 11	35 ± 8	34 ± 9	37 ± 13
Education level (%) post-secondary	0%	76%	100%	69%	36%

typically be found at fishermen's coves and recreational divers are easily located at dive centers, whereas scientific divers and spear fishermen are more difficult to reach. Consequently, our survey follows a convenience sampling strategy, meaning our sample is not necessarily representative of the total population in each group. However, a truly representative sample of each diver type would be impossible since there are no national-level data available on the participation rates of each diving activity.

RESULTS

Diver Characterization

Of the 229 divers interviewed, most are from Chile (95.2%) and male (82.5%), with an average age of 38.8 ± 11.3 years (**Table 2**). Compared to other diver types, recreational divers and instructors have a higher proportion of women with 32.7 and 23.5%, respectively. Artisanal fishermen are older than other diver types, with an average age of 47.9 ± 9.3 years; they also have the most basic educational level overall, generally with no post-secondary studies. Scientific divers have the highest education level, with all of them having completed post-secondary studies.

Diving Profile

The main diving mode of artisanal fishermen is diving with hookah (95.7%), and they focus mostly on the extraction of natural resources (91.4%). Instructors usually dive with SCUBA tanks (94.1%), and although their activities are mainly guiding and training (76.5%), a relatively high percentage (23.5%) also dive for their own recreation. Not surprisingly, most of the recreational divers (73.1%) dive with SCUBA tanks for recreation, and 19.2% also dive to take photos or videos. Scientific divers, who mainly dive with SCUBA tanks (72.4%), primarily dive for scientific work (40%), and almost 20% also take photos and videos. "Various" divers (64%) marked "other activity" as their main activity during a dive and almost 30% dive for resource extraction. Most of them (64.3%) are free divers.

Artisanal fishermen are the most experienced group with more years diving than other groups ($H = 104.4$; $p < 0.001$), while recreational divers are the least experienced (**Table 3**). The individual dives of artisanal fishermen and "various" divers are typically longer (averaging more than 200 min per dive) than those of the other diver types, being shortest for instructors ($H = 127.7$; $p < 0.001$). Despite the long duration, artisanal fishermen usually have no free time during their dives, while recreational divers have the most time available for potential scientific activities ($H = 98.7$; $p < 0.001$). Very few (12.7%) of the artisanal fishermen have an underwater camera or dive computer, while many of the instructors and scientific divers own and use these types of equipment (**Table 3**).

Interest, Preferences, and Motivation to Participate in CS

There is a generally high interest in participating in CS for all diver types (**Table 4**). All divers, regardless of type, express strong interests to learn about all scientific fields, knowledge, skills and all taxa (fishes, invertebrates, and seaweeds). Interest in data analysis is relatively low among all diver groups. The most preferred taxon to learn about is "fishes" and the lowest preference is for "marine mammals and seabirds." Artisanal fishermen show a relatively higher interest in mollusks and seaweeds than other diver types. All diver types express strong interest in receiving training even if they have to study and take an exam to participate in CS projects. The willingness to pay for training is relatively lower than to study or do an exam during the training, but still positive. Overall, the most preferred communication channels are e-mail and phone applications, but it is important to highlight that e-mail is not considered useful for artisanal fishermen, and neither is communication via a website. The remaining diver groups have high scores in all communication types, with the exception of letters through

TABLE 3 | Diving profile by diver type.

	Artisanal fishermen *N* = 71	Dive instructors *N* = 34	Scientific divers *N* = 58	Recreational divers *N* = 52	Various *N* = 14
Dive experience (years diving average ± SD)	31.4 ± 10.1 a***	14.7 ± 12.9 b**	15.0 ± 10.6 b**	6.6 ± 8.3 c***	22.1 ± 14.7 ab**
Dive duration (minutes average ± SD)	238.7 ± 100.4 a***	40.0 ± 18.7 c***	86.0 ± 75.0 b***	47.3 ± 36.8 bc***	202.1 ± 117.9 a**
Free time during dives^X (Categories average ± SD)	1.1 ± 0.7 c***	2.4 ± 1.5 b**	2.7 ± 1.7 b**	4.4 ± 1.2 a***	2.7 ± 2.0 bc**
Owns dive camera (%) yes	12.7%	82.3%	77.6%	69.2%	71.4%
Owns dive computer (%) yes	5.6%	79.4%	62.1%	38.5%	50.0%

*X: Ordinal categories from 1 to 4 where 1 = no free time, 2 = 5–10 min, 3 = 10–15 min, 4 = 15–30 min, 5 = more than 30 min. Coincidence in letters between diver types mean no significant differences between them. *Means differences with effect size between 0.8 and 1, ** represent differences with effect sizes >1 up to 2 and ***means differences with effect size bigger than 2 (see* ***Supplementary Table S1*** *for effect size values). The specific interview questions can be found in* ***Supplementary Data Sheet S1****.*

TABLE 4 | General interest and preferences in CS for each diver type.

		Total *N* = 229		Artisanal fishermen *N* = 71	Dive instructors *N* = 34	Scientific divers *N* = 58	Recreational divers *N* = 52	Various *N* = 14		
H	***p*-value**	***M* ± *SD***		***M* ± *SD***	***M* ± *SD***	***M* ± *SD***	***M* ± *SD***	***M* ± *SD***	***H***	***p*-value**
General Interest: I would be interested in. . ..										
		4.4 ± 0.9	GI1. participating in CS	4.2 ± 1.1	4.4 ± 0.8	4.5 ± 0.7	4.4 ± 0.9	4.2 ± 0.9	2.1	0.70
		4.3 ± 1.0	GI2. doing exclusive dives for CS	4.2 ± 1.1	4.3 ± 0.9	4.3 ± 1.0	4.5 ± 0.9	4.1 ± 1.3	2.2	0.69
Field Knowledge: I would like to be trained for. . .										
		4.4 ± 1.0	FK1. Species recognition	4.6 ± 0.8	4.47 ± 1.1	4.1 ± 1.1	4.4 ± 0.9	4.4 ± 1.0	9.7	0.04
7.07	<0.05	4.4 ± 1.0	FK3. Ecology (interactions between species)	4.4 ± 0.8	4.5 ± 0.8	4.3 ± 0.9	4.1 ± 0.8	4.8 ± 0.9	8.8	0.06
		4.2 ± 1.0	FK2. Biology (feeding and reproduction)	$4.5 \pm 0.9_{a}$	$4.2 \pm 0.9_{ab}$	$4.0 \pm 1.1_{b}$	$3.7 \pm 1.1_{b}$	$4.6 \pm 0.9_{ab}$	19.3	<0.01
Scientific Skills. I would like to be coached for. . .										
		4.0 ± 1.2	SS1. Scientific sampling protocols	3.9 ± 1.2	3.6 ± 1.3	4.2 ± 1.1	3.8 ± 1.2	4.64 ± 0.5	9.9	<0.05
11.6	<0.001	3.9 ± 1.3	SS2. Scientific experiments	4.1 ± 1.6	3.5 ± 2.1	4.1 ± 1.3	3.5 ± 1.6	4.6 ± 0.4	16.7	<0.01
		3.6 ± 1.4	SS3. Data analysis	$3.7 \pm 1.3_{ab}$	$3.0 \pm 1.5_{b*}$	$3.7 \pm 1.3_{ab}$	$3.3 \pm 1.3_{ab}$	$4.4 \pm 0.7_{a*}$	13.8	<0.01
Biodiversity. I would be interested to learn about...										
		4.4 ± 1.0	B1. Fishes	$4.5 \pm 0.8_{a}$	$4.5 \pm 1.0_{ab}$	$3.9 \pm 1.2_{b}$	$4.3 \pm 0.9_{ab}$	$4.7 \pm 0.8_{ab}$	17.6	<0.01
		4.3 ± 1.1	B2. Mollusks	$4.6 \pm 0.7_{a*}$	$4.4 \pm 1.0_{ab}$	$3.8 \pm 1.1_{b*}$	$4.1 \pm 1.0_{ab}$	$4.4 \pm 1.0_{ab}$	25.6	<0.05
		4.2 ± 1.1	B5. Crustaceans (crabs, shrimps. . .)	$4.5 \pm 0.9_{a}$	$4.3 \pm 1.1_{ab}$	$3.9 \pm 1.\ 1_{b}$	4.1 ± 1.1 ab	$4.5 \pm 0.9_{ab}$	15.2	<0.01
18.8	<0.001	4.2 ± 1.1	B7. Marine mammals and seabirds.	$3.8 \pm 1.2_{b}$	$4.7 \pm 1.1_{a}$	$4.3 \pm 1.1_{ab}$	$4.2 \pm 1.0_{ab}$	$4.6 \pm 0.7_{ab}$	21.9	<0.001
		4.1 ± 1.3	B5. Echinoderms (sea stars, sea urchins...)	4.3 ± 1.0	4.3 ± 1.1	3.8 ± 1.2	4.0 ± 1.2	4.0 ± 1.20	11.9	<0.05
		4.0 ± 1.5	B3. Cnidarian (corals, jellyfish, and sea anemones)	$3.8 \pm 1.3_{b}$	$4.5 \pm 0.7_{a}$	$3.8 \pm 1.2_{ab}$	$4.0 \pm 1.2_{ab}$	$4.1 \pm 1.2_{ab}$	11.0	<0.05
		4.0 ± 1.6	B6. Seaweeds	$4.4 \pm 1.1_{a*}$	$4.2 \pm 0.7_{ab}$	$3.8 \pm 1.0_{ab}$	$3.5 \pm 1.0_{b*}$	$4.1 \pm 0.5_{ab}$	16.3	<0.01
Participation in Training: I would be willing to. . .										
		4.2 ± 1.3	TP2. (. . .) be evaluated	4.5 ± 1.0	4.3 ± 1.1	4.1 ± 1.2	3.9 ± 1.1	4.7 ± 0.6	15.8	<0.01
113.7	<0.001	4.1 ± 1.2	TP1. (. . .) have to study	4.1 ± 1.4	4.2 ± 1.3	4.0 ± 1.2	4.0 ± 1.1	4.4 ± 0.6	2.9	0.5
		3.1 ± 1.8	TP3. (. . .) pay	3.0 ± 2.2	3.1 ± 1.3	2.8 ± 2.0	3.3 ± 1.4	3.3 ± 1.6	4.1	0.4
Communication: It would be useful to talk to scientists by. . ..										
		4.0 ± 1.3	C2. Phone	$4.4 \pm 1.0_{a*}$	$3.5 \pm 1.3_{b}$	$3.9 \pm 1.4_{ab}$	$3.4 \pm 1.2_{b*}$	$4.2 \pm 1.05_{ab}$	27.7	<0.01
		4.0 ± 1.5	C5. e-mail	$2.7 \pm 1.7_{b**}$	$4.8 \pm 0.7_{a*}$	$4.8 \pm 0.5_{a**}$	$4.5 \pm 0.87_{a**}$	$4.1 \pm 1.38_{ab}$	78.2	<0.001
352.5	<0.001	4.0 ± 1.3	C4. App	3.8 ± 1.4	4.0 ± 1.6	4.0 ± 1.29	4.0 ± 1.2	4.3 ± 0.84	3.2	0.5
		3.8 ± 1.4	C3. Web	$2.8 \pm 1.6_{b**}$	$4.2 \pm 1.1_{a**}$	$4.3 \pm 1.0_{a**}$	$4.3 \pm 1.0_{a**}$	$3.9 \pm 1.14_{ab}$	43.7	<0.001
		3.5 ± 1.5	C6. Social	$3.2 \pm 1.7_{b}$	$4.1 \pm 1.4_{a}$	$3.4 \pm 1.4_{ab}$	$3.4 \pm 1.47_{ab}$	$3.7 \pm 1.20_{ab}$	9.5	<0.05
		1.6 ± 1.1	C1. Letter	2.3 ± 1.4	1.1 ± 0.4	1.2 ± 0.8	1.2 ± 0.5	2.2 ± 1.31	7.0	<0.05

*Each question based on 1–5 point scale. Coincidence in letters between diver types mean no significant differences between them. *Means differences with effect size between 0.8 and 1, ** represent differences with effect sizes >1 up to 2 (see **Supplementary Table S1** for effect size values). The specific interview questions can be found in **Supplementary Data Sheet S1**.*

regular postal mail, which is the least preferred communication method for all divers.

The most preferred research protocol is "collecting samples" followed closely by "recording data after a dive" and "opportunistic observations" (**Table 5**). It is important to note that research protocols such as taking notes during a dive and conducting scientific transects and experiments had negative scores, e.g., are least preferred by all divers. In general, artisanal fishermen have lower scores for some protocols in which recreational and scientific divers score highest.

The main motivations to participate in CS are learning about the sea and contributing to science for all diver types (**Table 6**). Contributing to science is a slightly higher motivation for scientific divers. The motivation related to the enjoyment of the activity is higher for the recreational divers than for the artisanal fishermen. Social motivation (e.g., meeting people) is significantly lower for the recreational divers and dive instructors. The motivation with the lowest score is to obtain discounts for diving gear.

The PCA results highlight the main groups of divers based on all measured variables. The first two axes of the PCA explained 30.6% of the total variance (**Figure 2**). The PCA2 (11.8%) separates artisanal fishermen (on the positive side of the axis) from the other diver types (on the negative side of the axis). The main variables that distinguish artisanal fishermen from the other divers are the greater number of years diving and longer dive durations (**Figure 2**; DP5, DP4). Most recreational divers are on the extreme negative side of the axis, with the most important variables being their extensive free time during their dives, greater access to dive cameras and computers (DP6, DP7), preference for communication by web (C3), and preference for underwater protocols (**Figure 2**; PP4, PP5, and PP6).

DISCUSSION

This study distinguishes among different diver types based on their main diving activity, and clarifies some differences among the groups, which could influence their willingness to engage with and contribute to CS. All groups show a high level of interest in participating in CS projects and there are no strong differences in preferred ways of participation. However, strong differences in their diving activity could help to understand their needs or requirements to participate in CS projects. This information is useful for designing strategies to recruit a greater diversity of divers to participate in CS.

Characterization of Divers

In general, demographic characteristics found in the present study are consistent with the literature. For example, a general

TABLE 5 | Preferences in different CS protocols for each diver type.

		Total *N* = 229		Artisanal fishermen *N* = 71	Dive instructors *N* = 34	Scientific divers *N* = 58	Recreational divers *N* = 52	Various *N* = 14		
X^2	*p*-value	*M* ± *SD*	Protocols	*M* ± *SD*	*M* ± *SD*	*M* ± *SD*	*M* ± *SD*	*M* ± *SD*	*H*	*p*-value
		4.2 ± 1.7	P3. Collecting samples	3.8 ± 2.2	4.0 ± 1.6	4.1 ± 1.3	4.7 ± 1.3	4.5 ± 2.0	9.5	0.05
		3.8 ± 1.9	P2. Recording after a dive	$3.1 \pm 2.3_{b*}$	$3.9 \pm 1.6_{ab}$	$4.1 \pm 1.7_{ab}$	$4.2 \pm 1.6_{a*}$	$4.4 \pm 1.5_{ab}$	9.4	0.05
137.6	<0.001	3.2 ± 2.0	P1. Opportunistc observations	2.9 ± 2.3	4.1 ± 1.8	3.2 ± 2.0	3.0 ± 1.7	3.1 ± 1.9	8.0	0.09
		2.9 ± 1.9	P4. Notes while dives	$2.1 \pm 2.2_{b}$	$3.3 \pm 1.5_{ab}$	$3.3 \pm 1.5_{a}$	$3.3 \pm 1.5_{a}$	$2.4 \pm 1.6_{ab}$	14.9	<0.01
		2.7 ± 2.0	P5. Scientific transects	2.4 ± 2.3	2.5 ± 1.8	3.1 ± 1.9	2.8 ± 1.7	2.3 ± 1.9	4.9	0.3
		2.3 ± 1.9	P6. Scientific experiments	$1.5 \pm 1.8_{b}$	$2.5 \pm 1.7_{a}$	$2.8 \pm 1.9_{a}$	$2.5 \pm 1.6_{a}$	$2.3 \pm 2.1_{ab}$	20.4	<0.001

*Each question based on 1–5 point scale. Coincidence in letters between diver types mean no significant differences between them. *Means differences with effect size between 0.8 and 1 (see* ***Supplementary Table S1*** *for effect size values). The specific interview questions can be found in* ***Supplementary Data Sheet S1****.*

TABLE 6 | Motivations to participate in CS for each diver type.

		Total *N* = 229		Artisanal fishermen *N* = 71	Dive instructors *N* = 34	Scientific divers *N* = 52	Recreational divers *N* = 58	Various *N* = 14		
X^2	*p*-value	*M* ± *SD*	Motivation	*M* ± *SD*	*M* ± *SD*	*M* ± *SD*	*M* ± *SD*	*M* ± *SD*	*H*	*p*-value
		3.8 ± 1.1	M2. Learning about sea	4.0 ± 0.1	3.8 ± 01.0	3.8 ± 1.1	3.5 ± 1.1	3.6 ± 1.1	6.7	0.1
		3.6 ± 1.3	M3. Contribute to science	$3.4 \pm 1.3_{b}$	$3.7 \pm 1.1_{ab}$	$3.3 \pm 1.3_{b}$	$4.1 \pm 1.0_{a}$	$3.6 \pm 1.5_{ab}$	13.2	<0.05
219.6	<0.001	2.7 ± 1.3	M1. For fun	$2.3 \pm 1.3_{b}$	$2.8 \pm 1.3_{ab}$	$3.3 \pm 1.4_{a}$	$2.6 \pm 1.2_{ab}$	$2.2 \pm 1.4_{ab}$	18.0	<0.01
		2.7 ± 1.1	M4. Meeting people	$2.9 \pm 1.1_{ab}$	$2.3 \pm 1.0_{b**}$	$2.4 \pm 0.9_{b**}$	$2.8 \pm 1.1_{ab}$	$3.3 \pm 0.7_{a**}$	15.5	<0.05
		2.1 ± 1.6	M5. Obtaining discounts	2.3 ± 1.6	2.2 ± 1.7	2.0 ± 1.5	1.9 ± 1.5	2.3 ± 1.6	3.1	0.5

*Each question based on 1–5 point scale. Coincidence in letters between diver types mean no significant differences between them. *Means differences with effect size between 0.8 and 1, ** represent differences with effect sizes >1 up to 2 (see* ***Supplementary Table S1*** *for effect size values). The specific interview questions can be found in* ***Supplementary Data Sheet S1****.*

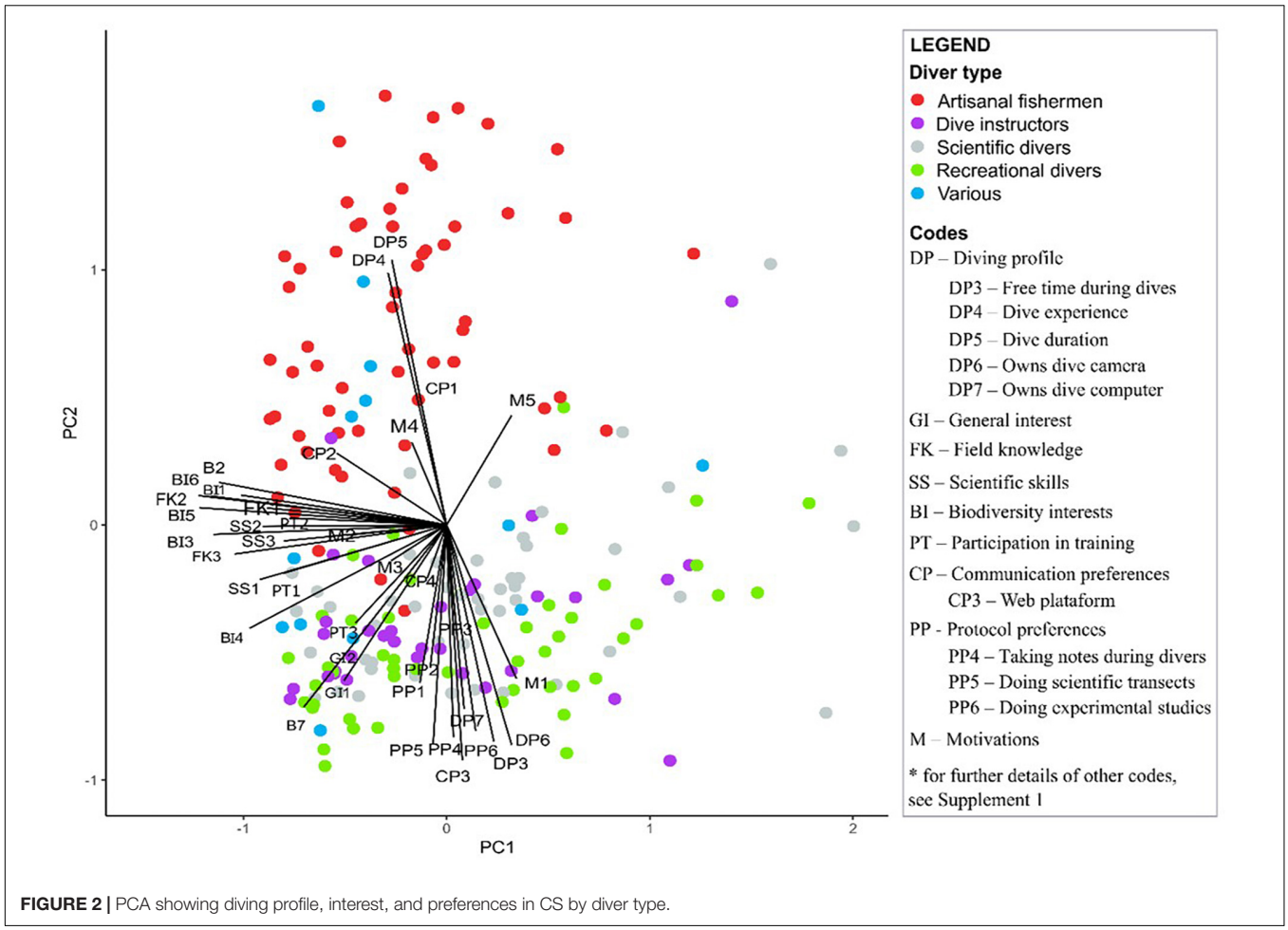


FIGURE 2 | PCA showing diving profile, interest, and preferences in CS by diver type.

70/30 ratio of male/female divers is frequently found in the literature for SCUBA divers (Lucrezi et al., 2018), and the much higher proportion of males among the artisanal fishermen is consistent with that found in other studies with fishermen (Thiel et al., 2014; de Andrade and de Oliveira Soares, 2017; de Juan et al., 2017). The average age for fishermen and instructors in this study is similar to that found in Curacao and Bonaire (Johnson and Jackson, 2015), and for recreational divers it is similar to other studies in diverse locations such as Spain, Malaysia, and Micronesia (Mundet and Ribera, 2001; Musa et al., 2006; Edney, 2012). A high educational level is also characteristic of SCUBA divers (Musa et al., 2006; Freiwald et al., 2018; Lucrezi et al., 2018), while a comparatively basic educational level has been previously documented for artisanal fishermen (Grant and Berkes, 2007; Carrasco and Menéndez, 2013; de Andrade and de Oliveira Soares, 2017).

Differences in diving experience among groups have also been reported before. For example, recreational divers are generally less experienced than other divers or fishermen (Johnson and Jackson, 2015; Tessier et al., 2015). The number of years diving in recreational divers (6.6 ± 8 years) is similar to that found in Australia (7 ± 9 years; Hammerton, 2017), Miami (median 6.5; Stang and Wiener, 1970) or Micronesia where most of the divers have been diving for less than 15 years (Edney, 2012), and slightly lower than in Barbados (10.7 ± 9.6; Kirkbride-Smith et al., 2013). Our study simplifies diving experience using only the number of years of diving experience, which might be a limited way to characterize the complexity of "experience." Other studies include more variables, such as total dives logged or number of diving certifications (e.g., Lucrezi et al., 2013; Cerrano et al., 2016). However, most artisanal fishermen have only one single dive license (the one they are legally required to have), and they usually do not keep a dive log, yet of all diver types, they spend the greatest amount of time underwater, accumulating extensive experience. Thus, while other variables might be useful, within the Chilean context the total years diving is considered the most useful indicator for diving experience.

Dive duration in our study for recreational divers (47.3 ± 36.8 min) is slightly lower than reported for Philippines (56 ± 6.4 min; Vianna et al., 2014), Florida (54.2 min; Camp and Fraser, 2012), or in the Red Sea at Sharm el Sheik (48.6 min; Branchini et al., 2015), and more similar to that found in the Mediterranean Sea (44–48 min; Terrón-Sigler et al., 2016). The shorter dive time in our study might be explained by the comparatively low water temperature in the Humboldt Current System (Thiel et al., 2007), which naturally reduces the amount

of time divers are able to spend in the water (Stang and Wiener, 1970). No previous study has investigated the amount of free time during dives, which is important information to take into account when developing CS projects that rely on volunteers.

Motivations and Preferences to Participate in CS

The results of this study show that contributing to science and to personal learning are typical motivations to participate in CS projects, irrespective of the type of diver. Similar motivations have been reported in other studies with divers (Cerrano et al., 2016; Martin et al., 2016a; Lucrezi et al., 2018). Meeting people is of less importance for instructors and recreational divers than for other diver types. Both diver types are involved in diving as a social activity, with meeting people being an inherent part of the activity (MacCarthy et al., 2006; Dimmock, 2009), which is why they might place less emphasis on this aspect. Understanding motivations is useful for proposing incentives strategies for CS projects to engage divers. Based on our results, it is important to all diver types that CS projects have good outreach instruments to share the scientific results. Feedback to participants has also been found to be an important component of public engagement in marine CS (Martin et al., 2016b).

The preferences divers expressed for CS protocols indicate that divers are not equally interested in performing the different tasks during their dives. In general, more complex tasks in CS projects tend to attract fewer volunteers or limit their participation (Bonney et al., 2009; Franzoni and Sauermann, 2014). Many potential volunteers think they need more knowledge or skills to contribute to CS projects, which might discourage them from participating (Martin et al., 2016a; Lucrezi et al., 2018). Even though some artisanal fishermen mentioned that they have experience in helping professional scientists with underwater transects, they were the group with the lowest interest in using transects. This could be explained by the very limited free time during their dives. Therefore, one recommendation for initiating a project in a location where divers have limited CS experience is to start with less complex protocols, even when some divers have extensive diving experience.

Divers in this study expressed a strong interest in participating in CS projects. This finding is similar to other studies of divers (Martin et al., 2016b; Lucrezi et al., 2018). Most CS projects with divers are in the field of biology and ecology (Thiel et al., 2014) and their interest in these two fields is reflected in our results. Divers, in general, are less interested in participating in data analysis, even though the average scores were still positive. Martin et al. (2016b) found that divers are more interested in data analysis than other marine users, which is likely due to the high science education level of the divers surveyed in that study. On the other hand, artisanal fishermen are very interested in research on mollusks and seaweeds, which appears reasonable since these organisms are one of the main target resources they harvest (Aburto et al., 2013). The participation of fishermen in CS has been related to their fishing target species (e.g., Obura, 2001; Le Fur et al., 2011), but there are also successful experiences with other taxa (e.g., Aswani and Lauer, 2006; Azzurro and Bariche, 2017), which herein is reflected in their wide interest in diverse taxonomic groups.

Communication channels in CS are very important to maintain volunteers' engagement with CS projects (Cooper et al., 2007; Tulloch et al., 2013). All diver types in this study find electronic communication useful with the exception of artisanal fishermen, for whom e-mail and the internet is not a useful method. Artisanal fishermen in Chile usually live in remote fishing villages and have restricted access to the internet (Gallardo et al., 2011), yet our study found they have positive scores in other communication channels that require internet access (such as mobile applications and social networks).

The Diversity of Divers and Designing CS Projects

Different diver types have similar preferences and motivations to participate in CS, but there are important differences in their profile that could affect their participation in CS projects. For example, the differences in education level could influence their decision to participate, since a low level of education has been reported as a barrier to CS projects (Savio et al., 2017) and can affect data quality (Delaney et al., 2008). Our findings show that artisanal fishermen, despite their comparatively basic science literacy, have a high level of diving experience. This means they will likely have better diving skills and more knowledge about species and ecosystem changes (Macdonald et al., 2014), which should positively affect data quality. They are familiar with local species and spend long hours underwater, making them important collaborators for studying changes over time, by contributing their deep local ecological knowledge (LEK) (Drew, 2005; Stephenson et al., 2016). Furthermore, they usually live near the shore, and observe their local marine habitats continuously, including during unfavorable weather conditions, such as winter seasons or storm events. These characteristics make fishermen ideal partners in a monitoring system that aims at detecting shifts in the ecosystem, including changing species abundances or arrival of exotic species. Their dependence on marine resources (de Juan et al., 2017) and their extensive knowledge about the marine environment means they could contribute to new, relevant research questions. However, to facilitate their participation in a CS project, it is important to keep in mind that fishermen prefer face-to-face conversations to share their experiences rather than responding to anonymous surveys or completing complex protocols (Obura et al., 2002; Thiel et al., 2014).

Recreational divers on the other hand are highly educated (Martin et al., 2016b; Freiwald et al., 2018) and have most of their dive time available for making observations or scientific activities, but have less diving experience. Some CS projects require intensive activities during a dive, such as conducting scientific transects (Edgar et al., 2014; Roelfsema et al., 2016). For these kinds of projects, a minimum diving experience and training is required (Edgar et al., 2014; Roelfsema et al., 2016), which reduces the pool of potential volunteers. Furthermore, transect protocols are incompatible with most dives conducted by a diving center. To overcome this concern, some projects

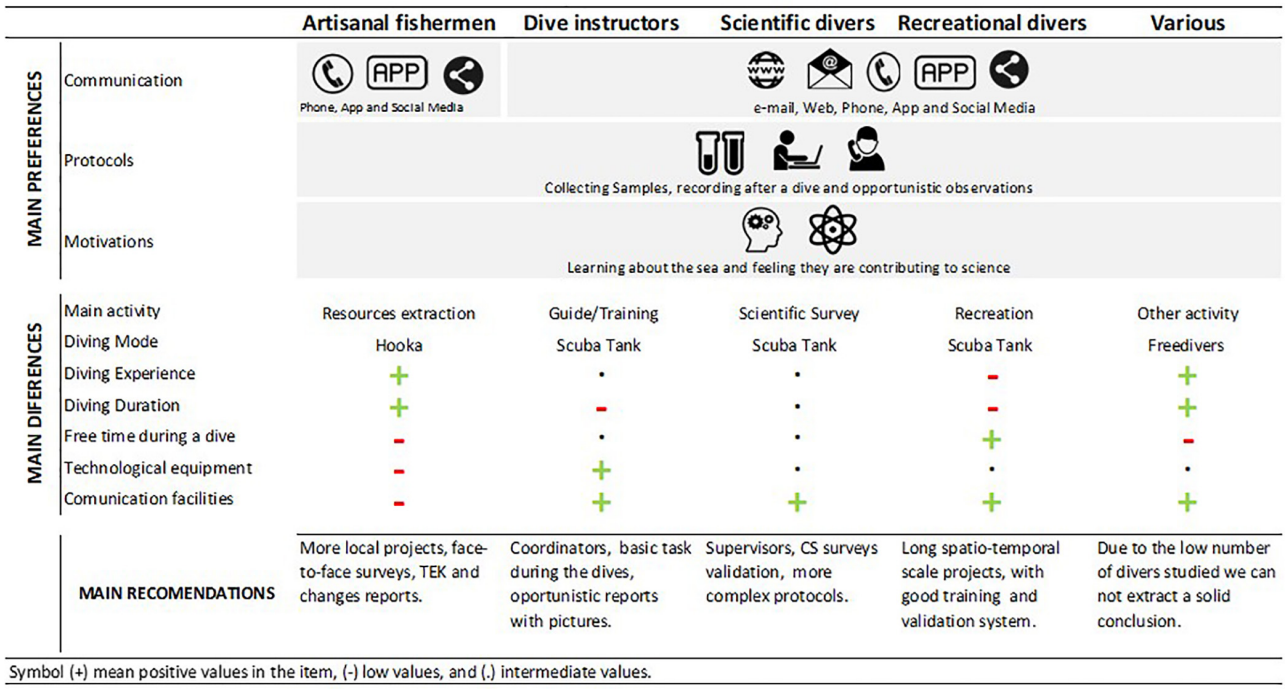


FIGURE 3 | Main preferences and differences between diver types and recommendations for each, based on the results of the present study.

have developed simplified data recording protocols (Bonney et al., 2009; Edgar and Stuart-Smith, 2009; Freitag et al., 2016; Vermeiren et al., 2016), for example, those based on opportunistic observations (Huveneers et al., 2009; Couturier et al., 2015; Ward-Paige et al., 2018).

Some CS projects with divers are using photographs, e.g., iNaturalist (Fourcade, 2016) or REDMAP (Pecl et al., 2014). This could represent a barrier for artisanal fishermen as they generally do not have an underwater camera, while dive instructors, on the other hand, would fit very well with this type of project as their dives are usually long, and most of them own cameras. Furthermore, they are familiar with (and often own) dive computers, which provide more accurate data about the physical conditions during the dives (such as time, depth, temperature; Wright et al., 2016).

Based on the above considerations it is possible to express several simple recommendations for CS projects for all types of divers (**Figure 3**). We recommend artisanal fishermen for more local projects where it is possible to engage in face-to-face interactions, contractual, or co-created CS projects (Shirk et al., 2012). This will be particularly valuable for projects aiming to detect ecosystem changes. Recreational divers, who can be contacted via social media or other online communication tools, might be best engaged in large-scale, contributory projects (Shirk et al., 2012). However, due to their low diving experience and irregular contact with marine ecosystems, it will be very

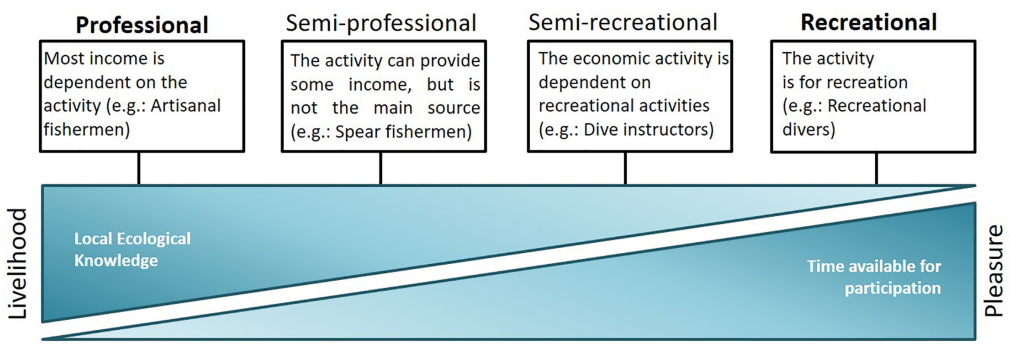


FIGURE 4 | Differences between potential volunteers in CS based on their economic dependence on the activity related to a CS project.

important to create good validation systems for these CS projects. If long-term engagement is achieved with recreational divers, they may improve their skills over time. For this reason, recreational diver projects should consider starting out with more simple tasks (e.g., reporting photographic records) and advancing toward more complex tasks (including specific samplings or transect protocols). At the same time, considering the extensive diving experience of the instructors, their general interest in CS and their direct interaction with recreational divers (Lucrezi et al., 2018), we recommend them as a strategic figure in CS projects with recreational divers. In addition, scientific divers would be appropriate to generate trust among other diver types, explain protocols and take part in a data validation system. For "various" divers, due to our lack of data, we propose to develop more studies that focus on spear fishermen, because this group in particular can help to contribute to the documentation of long-term shifts in resource traits (Godoy et al., 2010; Young et al., 2015).

CONCLUSION

The results highlight the importance of knowing the characteristics of different diver types to determine the best way of recruiting them to CS projects. We found all diver types have a generally high interest in CS, along with similar preferences for how they participate. However, there are some important differences in their diving profile that are likely to determine how they will and can participate in CS. The strongest contrast became evident between artisanal fishermen and recreational divers, for whom the nature of their underwater activity and equipment has a direct bearing on their capability to record observations for CS. Thus, categorizing potential volunteers as professionals (who contribute to a project during their professional activity) and recreational (who contribute to a project from their hobby activity) could be useful to understand differences in volunteer preferences and requirements to participate in CS projects. In general, it is expected that professional volunteers have a high potential to collect valuable information about the environment based on their LEK, but they have specific requirements due to their lack of free time during their professional activities. In contrast, recreational volunteers will have another specific potential (more time to participate) but may have much more limited ecological knowledge, meaning there is a limit to the complexity that should be involved in the data collection process for this group. The current results on different diver types may extend to different types of marine users (e.g., cargo shipping, fish trawlers, coast guards, or recreational captains) or beach users (e.g., lifeguards or tourist visitors). Knowing the interests of potential volunteers and what is required to engage them will improve the design of CS projects and effective recruitment and retention of volunteers. Identifying and grouping potential volunteers with similar preferences and requirements will help to design more engaging projects (**Figure 4**) for other groups of volunteers participating in marine (or even terrestrial) CS.

ETHICS STATEMENT

This study a carried out in accordance with the recommendations of "Reglamento del Comité Ético Científico, Comité Ético Científico de la Sede de Coquimbo de la Universidad Católica del Norte," with written informed consent from all subjects. All subjects gave written informed consent in accordance with the Declaration of Helsinki. The protocol was approved by the "Comité Ético Científico de la Sede de Coquimbo de la Universidad Católica del Norte under resolution N° 2017-6".

AUTHOR CONTRIBUTIONS

MH, MT, and WS contributed conception and design of the study. MH, MT, WS, and SG performed the methodology. MH organized the database and wrote the first draft of the manuscript. MH and VM performed the statistical analysis. VM, MT, WS, and SG reviewed and edited the manuscript. All authors contributed to manuscript revision, read, and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00239/full#supplementary-material

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POLICY BRIEF
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Identifying Priorities for Shark Conservation in the Bay of Bengal, Bangladesh

Mahatub Khan Badhon[1], Md Kutub Uddin[2], Fahmida Khalique Nitu[3] and Enamul Mazid Khan Siddique[4]*

[1] Department of Zoology, University of Dhaka, Dhaka, Bangladesh, [2] Voices for Interactive Choice and Empowerment (VOICE), Dhaka, Bangladesh, [3] National University of Bangladesh, Gazipur, Bangladesh, [4] Bangladesh Youth Environmental Initiative (BYEI), Dhaka, Bangladesh

Fisheries bycatch is known as the major threat to Threatened shark species (herein, sharks, skates, and rays) in Bangladesh. But bycatch is not appropriately addressed under the existing wildlife and fisheries conservation management regime. This policy brief evaluates the current scenario of shark conservation and identifies priorities for future interventions. The literature review finds 71 shark species and only four peer-reviewed publications from Bangladesh suggesting the species already known have not yet been studied. In addition, inconsistencies in legal frameworks have limited the capacities and mandates of responsible government agencies. We recommend actionable changes in policy to regulate shark trade, reduce bycatch of Threatened species, improve fisheries data reporting system, and bring consistency between institutional mandate and the capacity of conservation and management agencies.

Keywords: sharks, threatened species, conservation priorities, sustainable fisheries, bycatch, Bay of Bengal, Bangladesh

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****Correspondence:***
Mahatub Khan Badhon
mahatub.badhon@du.ac.bd

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INTRODUCTION

Conservation efforts to protect shark species (herein, sharks, skates, and rays) are of paramount importance. The IUCN Red List's Shark Specialist Group (Dulvy et al., 2014) estimates that 24% of all extant sharks and chimeras are threatened with an elevated risk of extinction. The utility of shark products is driving targeted and opportunistic shark fishing (Lack and Sant, 2009; Dent and Clarke, 2015). Exposure to fishing mortalities (Bonfil, 1994) and low intrinsic rates of population growth (Musick et al., 2000; Frisk et al., 2001) mean there is a critical need for effective shark conservation measures.

The extensive fishing pressure in the Bay of Bengal (Khan et al., 1997) drives the catch of shark species in Bangladesh. Coastal and marine fisheries operate 67,669 artisanal boats and 253 commercial trawlers (Department of Fisheries [DOF], 2018), and support 2.7 million people (Program Development Office for Integrated Coastal Zone Management Plan [PDO-ICZMP], 2003). The fishing fleet uses the depth zone between 10 and 80 m (Chowdhury, 2017), with fisheries surveys conducted by Hida and Pereyra (1966) and Sœtre (1981) concluding that the average proportion of shark catch is higher in this depth zone than in other depths.

Sharks are predominantly taken as bycatch (Haldar, 2010; Hoq, 2010), and among those caught are species listed in the Threatened category (assessed as Critically Endangered, Endangered, or Vulnerable) by IUCN Red List. Targeted and opportunistic shark fisheries also exist in Bangladesh's

water (Bahadur, 2010; Haque et al., 2018). Processing plants in coastal areas prepare export-oriented shark products, where no part of a shark is discarded (Bahadur, 2010; Haque et al., 2018). No local demand for shark fin products has been reported (Hasan et al., 2017; Haque et al., 2018), and only a few small communities of ethnic minorities consume fresh and dried shark meat (Bangladesh Bureau of Statistics [BBS], 2003; Roy et al., 2011; Haque et al., 2018). The evidence is inconclusive on whether only market demand is driving shark fishing (Hoq, 2010), but all types of fishing in coastal and marine areas have Threatened shark species as bycatch.

As part of global efforts in advocating shark conservation (Simpfendorfer et al., 2011; Dulvy et al., 2014, 2017; Shiffman and Hammerschlag, 2016a,b), the above-mentioned context has encouraged us to conduct this review to understand the conservation scenario for sharks, and to identify priorities for intervention in Bangladesh. Here, we recommend changes in policies and practices that can be introduced to reduce bycatch of Threatened shark species and mainstreaming shark conservation as an everyday part of Bangladesh's current fisheries management regime.

METHODS

A literature review and key informant (officials from the Department of fisheries, the Forest Department, independent researchers, conservation practitioners, local traders, exporters, and artisanal fisherfolk, and crews of commercial trawlers) interviews were conducted. For the literature review, we searched peer-reviewed articles published between 1970 and 2018 in the ISI Web of Science database using two set of keywords, i.e., "(Shark* OR Chondrichthy* OR Elasmo*) AND Bangla*" and "(Shark* OR Chondrichthy* OR Elasmo*) AND Bengal*." The keywords were selected to search all papers related to shark species in the Bay of Bengal, Bangladesh. Out of 15 papers resulting from searches, we found four relevant papers; this number is extremely low compared to better-studied parts of the ocean. To find gray literature, non-systematic search queries were made in Google Scholar using the above-mentioned keywords in different combinations. In total, we eventually reviewed four peer-reviewed papers and other 26 documents, reports, and working papers, and used the framework for biodiversity knowledge shortfalls, reviewed and further developed by Hortal et al. (2015), to determine knowledge shortfalls on sharks.

Before interviewing key informants, we conducted 15 reconnaissance surveys (brief surveys to collect preliminary information on shark fishing, landing, and trading; to identify and locate key informants; and to look for issues, problems, and opportunities not mentioned in literature) in coastal areas since January 2015. These surveys allowed us to identify key informants. Given the legal implications and sensitive nature of shark trade, it took time to build up relationships with key personnel using snowball sampling. At the end of 2017, we used a semi-structured questionnaire (**Supplementary Material A**) to interview 20 individuals. Given the relatively small sample size, we also used informed participants' observation and perspectives as evidence.

RESULTS

Knowledge Shortfalls on Sharks

The review showed limited taxonomic description and cataloging (Brown and Lomolino, 1998) of sharks in terms of integrating molecular techniques with classical morphological systematics to resolve the taxonomy of shark species, as suggested by Last (2007) and White and Last (2012). The total number of named shark species occurring in Bangladesh is difficult to substantiate as credible taxonomic identification and qualified taxonomists are lacking. The number ranges from 22 to 56 (Hussain, 1970; IUCN, 2000; Roy et al., 2007). Compiling Krajangdara et al. (2008), Haroon's (2011) figures, and Hoq and Haroon (2014), the total number of reported shark species reaches 71 (**Supplementary Material B**). Concerns over accurate taxonomic identification of shark species occurring in Bangladesh have been reported (Bay of Bengal Large Marine Ecosystem Project [BOBLME], 2011; Roy et al., 2014) previously, and the proposed National Plan of Action for shark (NPOA-shark) (Haldar, 2010) recognizes this discrepancy on the reported number of species in different literatures.

Taxonomic study provides a critical baseline and functional unit for biological research. Furthermore, one needs to define and enumerate species before developing species-specific conservation strategies. Simpfendorfer et al. (2011) have identified taxonomic research as one of the key conservation research priorities for sharks. We found one paper on *Glyphis gangeticus* (Roberts, 2006) and one taxonomic guidebook on sharks (Hoq and Haroon, 2014) that exclusively contribute to taxonomic research; however, none employed molecular approaches for species identification. Credible taxonomic diagnosis of shark species entails a meticulous examination of morphological systematics, often accompanied by molecular analysis, to eliminate confusion and misidentification of hybrids and species complexes. White and Last (2012) used the case of *G. gangeticus* occurring in the Bay of Bengal claimed by Roberts (2006) as an example to showcase that inadequate taxonomic investigation has the potential to mislead and create confusion.

We found three studies (Roberts, 2006; Rowat et al., 2008; Hossain et al., 2015) that employed historical and oral data from fisherfolk to understand the distribution of sharks. The survey effort and coverage of majority of the studies have been mostly limited to the fish markets and fish landing stations of the southeastern coast (Roy et al., 2007; Karim et al., 2012; Hoq et al., 2012; Hoq and Haroon, 2014). No offshore survey has been conducted to exclusively understand the spatial distribution, movement patterns and migratory routes of sharks. This ultimately results into an inadequate understanding of the geographic distribution of sharks (Lomolino, 2004).

The necessity of research on taxonomy, stock assessment, life history, biology and sustainable utilization of sharks has been suggested by Rahman and Uddin (2010) and Fischer et al. (2012). However, research to minimize the lack of data on

shark species remains significant in Bangladesh. The current unevenness in survey efforts (i.e., lack of species-specific catch data and poor sampling strategy) and coverage (i.e., geographic bias) could result in variation in the quality and reliability of the data available for future conservation planning (Gaston and Rodrigues, 2003; Mace, 2004).

Conservation Management Regime

We have identified 8 instruments that together build the legal and policy framework related to sharks in Bangladesh, namely, Forest Act, 1927, Protection and Conservation of Fish Act, 1950, Marine Fisheries Ordinance, 1983, National Fisheries Policy, 1998, Wildlife (Conservation and Security) Act, 2012, Bangladesh Biodiversity Act, 2017, the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), and the Convention on the Conservation of Migratory Species of Wild Animals (CMS). The Department of Fisheries and the Forest Department are the authorized agencies under this framework. The Forest Department is authorized to implement Wildlife (Conservation and Security) Act, 2012 and Forest Act, 1927. It is also the designated agency to represent Bangladesh in CITES and to work as the national authority to issue import and export permits under CITES. Wildlife Act (Government of Bangladesh [GoB], 2012) listed 29 species of sharks as protected; among these protected species, 15 are not listed in the Threatened criteria of IUCN Red List, and there was no national assessment before listing them as protected. Multiple key informants informed that the law is ineffective and poorly implemented because the mandate[1] of Forest Department is not consistent with its responsibility. Under this law, traditionally the Forest Department had species with different types of protection which inhabit terrestrial forest or wetlands but marine areas or fisheries were not part of Forest Department's mandate. Fisheries' bycatch is the main threat to sharks and the Forest Department does not have any mandate to manage marine fisheries thus rendering the protected status of sharks under the Forest Department obsolete.

Convention on International Trade in Endangered Species is a driver to improve the management of listed sharks (Vincent et al., 2014). According to key informants, Bangladesh, as a party to CITES, has not introduced any monitoring or management mechanisms to regulate the trade of shark products. CITES functions as an additional measure to regulate trade of shark products by restricting or controlling the international trade of a limited number of shark species as listed in the Appendix I and II lists (**Table 1**). The latest evidence detects species from CITES Appendix I (*Pristis pristis*) and Appendix II (*Sphyrna lewini*, *Alopias* sp., *Rhincodon typus*, and *Mobula japanica*) at fish processing plants in Bangladesh (Haque et al., 2018). It suggests that both undocumented trade and lack of monitoring on CITES-listed sharks are underway in Bangladesh. All CITES Appendix II listed species need to have non-detriment findings (NDFs) made for them to ensure that the numbers being removed are sustainable (Vincent et al., 2014). Despite being a range country for nine shark species listed in CITES Appendix II, Bangladesh is yet to prepare any NDFs. This shows the unpreparedness (i.e., lack of resources, infrastructure, and expertise) of the regulatory agencies to manage fisheries since the requirements for NDFs and sustainable fishery management goals at the domestic level are the same (Cochrane and Doulman, 2005). The Forest Department—being the management and enforcement authority of CITES in Bangladesh—has no mandate to manage the fisheries resources of Bangladesh.

Preparing NDFs is not only about setting sustainable quotas, but also about enforcing better bycatch regulations to reduce bycatch of Threatened or protected species, or increase post-release survival rates from non-selective gears (Vincent et al., 2014). Owing to the nature of multi-species fisheries and the absence of species-specific shark catch data (Mundy-Taylor and Crook, 2013), commercial capture of species listed in CITES Appendix II is currently difficult to substantiate. It is even more difficult to inform NDFs. Key informants suggest that resource mobilization, capacity building and collaboration between regulatory agencies (FD and DOF) are required to monitor trade (Appendix I and II) and investigate existing or potential fisheries for shark species (Appendix II). Most of the key informants agreed that if the demand decreases or authorities regulate the trade it will help to reduce the targeted and opportunistic shark fishing, but bycatch mortalities will continue. One key informant said that multi-species fisheries with gill nets in the coastal waters of Bangladesh are key contributors to bycatch mortalities of Threatened sharks.

Bangladesh ratified the Convention on the Conservation of Migratory Species of Wild Animals (CMS) and became a party to the treaty in December 2005. CMS party states have signed into effect the Memorandum of Understanding on the Conservation of Migratory Sharks (CMS Sharks-MoU) in March 2010; the first such global instrument on shark conservation. But, Bangladesh is yet to sign the CMS Sharks-MoU.

The marine fisheries sector in Bangladesh is shaped by laws and policies namely, Protection and Conservation of Fish Act, 1950, National Fisheries Policy, 1998, and Marine Fisheries Ordinance, 1983 focusing on increasing the catch of a few commercially valuable species (e.g., *Tenualosa ilisha*, a herring-like species) through restricting the use of specific fishing gears or delimiting fishing at different spatial and temporal scales (Islam et al., 2016). As the authorized agency under these laws, the Department of Fisheries is not mandated to conserve marine megafauna, the marine turtle is the only exception. Despite legislation on installing turtle excluder devices (TED) in trawl nets (Marine Fisheries Rules, Section 14A) in order to mitigate bycatch mortalities of turtles from commercial trawling, key informants from the trawling industry informed that no trawlers comply with TED regulation. The stringent oversight on commercially valuable fisheries management (Islam et al., 2016), in contrast to no enforcement on TED installment and no bycatch regulations of Threatened sharks, suggests that the prospect of revenue trumps conservation priorities in the current fisheries management regime.

[1] http://www.bforest.gov.bd/site/page/b24cdba5-14e0-4fde-8114-b7a557038915/-

TABLE 1 | List of shark species occurring in Bangladesh that have been listed in the Appendixes of CITES and CMS.

Species	Common Name	IUCN Red List Status	CITES Status	CMS Status
Rhincodon typus	Whale Shark	Endangered (2016)	Appendix II (2003)	Appendix I (2017)
Sphyrna lewini	Scalloped Hammerhead	Endangered (2007)	Appendix II (2013)	Appendix II (2014)
Sphyrna mokarran	Great Hammerhead	Endangered (2007)	Appendix II (2013)	Appendix II (2014)
Sphyrna zygaena	Scalloped Hammerhead	Vulnerable (2005)	Appendix II (2013)	Not Listed
Carcharhinus falciformis	Silky Shark	Near Threatened (2016)	Appendix II (2017)	Appendix II (2014)
Mobula kuhlii	Shortfin Devil Ray	Data Deficient (2009)	Appendix II (2017)	Appendix I (2014)
Mobula mobular	Giant Devil Ray	Endangered (2015)	Appendix II (2017)	Appendix I (2014)
Mobula japanica	Spinetail Devil Ray/ Spinetail Mobula	Near Threatened (2006)	Appendix II (2017)	Appendix I (2014)
Anoxypristis cuspidata	Narrow Sawfish	Endangered (2013)	Appendix I (2007)	Appendix I (2014)
Pristis pristis	Largetooth Sawfish	Critically Endangered (2013)	Appendix I (2007)	Appendix I (2014)
Alopias sp.	Thresher Shark	Vulnerable (2009)	Appendix II (2017)	Appendix II (2014)

There were regional efforts, for instance, the Bay of Bengal Large Marine Ecosystem (BOBLME) project (2008 to 2013)[2] of the Food and Agriculture Organization of the United Nations contributed to efforts to integrate conservation of marine megafauna in fisheries sector. But as these kinds of top-down global or regional efforts are not driven by demand of national institutions, after such projects end, national institutions do not internalize the process and do not take the ownership of outcomes. For instance, the Department of Fisheries was the national implementing agency of BOBLME, but it did not officially adopt a National Plan of Action for Sharks (NPOA-sharks) prepared under this project. Key informants have identified lack of resources as one of the key reasons behind this. After the project ended, Department of Fisheries did not allocate any resources to work on adopting NPOA-sharks, they said.

The newly introduced Bangladesh Biodiversity Act, 2017 (Government of Bangladesh [GoB], 2017) has the provisions needed to be the basic legislation for wildlife conservation. No single government ministry or agency is handed down the authority to implement this law. Rather, a multi-agency national committee is authorized to work with a multi-sectoral approach to conserve biodiversity and sustainable use of its resources. There are provisions for determining and protecting endangered species under this law from which Threatened shark species can benefit.

ACTIONABLE RECOMMENDATIONS

The government agencies, academia, and local and international conservation groups working on marine megafauna conservation in the Bay of Bengal should prioritize accurate taxonomic identification of shark species occurring in Bangladesh's waters. The discrepancy on the reported number of species must be resolved, there should be a national register of reported shark species. To develop the critical baseline and functional units for biological research on sharks in Bangladesh, we recommend that the government should facilitate long-term studies through its agencies, such as the Department of Fisheries and the Bangladesh Fisheries Research Institute. These agencies should host a consortium of relevant experts who will work on taxonomic identification, distribution, and population of sharks to build evidence that will guide the conservation and management of sharks in Bangladesh.

Department of Fisheries should finalize and adopt the NPOA-Sharks; if needed it should seek in-country technical assistance from UN-FAO as outlined in the International Plan of Action for Conservation and Management of Sharks (IPOA-Sharks).

To create evidence, acquiring species-specific data from different fish landing stations is critical because these data could help to initiate the process of national assessment on the status of shark species. We recommend the inclusion of new data attributes in the fisheries resources survey system (FRSS) to ensure species-specific records of shark species. As FRSS is a long-established mechanism under the Department of Fisheries, inclusion of species-specific assessment of shark landing will not require new resources. We strongly recommend that the Forest Department should start facilitating the process to prepare NDF to regulate the trade of shark products. Building capacity and infrastructure for the Forest Department to identify species from shark products and monitor trade for a CITES-compliant trade regime should also be a priority. As the management and regulatory agency of the fisheries sector, the Department of Fisheries should be given the technical responsibility to prepare the NDF.

Bangladesh should sign the CMS Sharks-MoU; it has policy mechanisms to protect highly migratory sharks by prohibiting take (of Appendix I species) or by requiring nations to cooperate on regional management (of Appendix II species) (McClenachan et al., 2012). Also, the MoU will provide Bangladesh with the opportunity to easily develop policy process that could address fisheries bycatch of whale sharks and sawfishes (Hossain et al., 2015; Adnan et al., 2018).

Conservation of sharks should be mainstreamed into fisheries policies and management; existing policy and legal instruments have scopes to do that. Article 8 and 8.2 of the National Fisheries Policy[3] should incorporate clear provisions outlining how it will

[2]https://www.boblme.org/project_document.html

[3]https://mofl.gov.bd/sites/default/files/files/mofl.portal.gov.bd/policies/13c01764_17a9_40ba_a170_ec758a651724/Jatio%20Matshya%20Niteemala.pdf

reshape legal processes in marine fishing industry to conserve Threatened shark species. It should also include directions about introducing bycatch regulations related to sharks and other marine megafauna. Importance should be given on greater investment in strategies to manage bycatch (i.e., modifications of gear, safe handling and releasing bycatch, reducing post-release mortalities) in artisanal and industrial fisheries.

The multi-agency national committee under the Bangladesh Biodiversity Act should start a national assessment of status of shark species. Species found as nationally Threatened should be protected under the Protection and Conservation of Fish Act. We recommend the Department of Fisheries, as the authorized agency to implement the Fish Act, should be allocated the necessary resources for enhancing institutional capacity and training of its human resources to engage the fishing industry for the protection of sharks and other marine mega fauna.

CONCLUSION

Achieving sustainable outcomes for most or all shark populations requires species-specific identification and understanding of the fisheries in context of a given geographic area (Dulvy et al., 2017; Simpfendorfer and Dulvy, 2017). Based on the paucity of species-specific research and data on sharks, and the poor state of conservation management, we have recommended a mix of priority actions that can transform the mutually exclusive and single sector approach of regulatory agencies (i.e., DOF and FD) to become more integrated. This national preparedness may set a strong base for implementing international conventions like CITES and CMS to regulate the trade of shark products and reduce bycatch.

ETHICS STATEMENT

We took written informed consent before audio recording all interviews. The Ethical Review Committee of the Faculty of Biological Sciences, University of Dhaka (72/Biol.Scs./2018–2019), approved the protocol and the approach of this study.

AUTHOR CONTRIBUTIONS

EMKS and MKU planned and designed the policy brief. FKN, MKB, and MKU conducted the literature review, reconnaissance surveys, and Key Informant Interviews. MKB and MKU developed the manuscript. All authors provided critical feedback.

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SUPPLEMENTARY MATERIAL

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OPINION
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The Need for a Code of Professional Ethics for Marine Conservation Communicators

Lucy E. Erickson[1], Sally Snow[2], Md. Kutub Uddin[3] and Gianna M. Savoie[4]*

[1] Marine Stewardship Council, London, United Kingdom, [2] Large Marine Vertebrates Research Institute Philippines, Bohol, Philippines, [3] VOICE, Dhaka, Bangladesh, [4] Ocean Media Institute, Bozeman, MT, United States

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As a crisis sector, marine conservation needs continuous public scrutiny to maintain much-needed transparency, accountability, and to secure public trust. Such opportunities for public scrutiny can be ensured through independent, objective and critical journalism (Johns and Jacquet, 2018). However, mainstream media and other journalistic platforms often rely on communication professionals working at marine conservation groups for information and expertise related to marine conservation issues. It is therefore crucial that communication professionals at conservation groups have a professional code of conduct that encourages dissemination of objective truth about conservation efforts and does not prevent journalists from carrying out their duties to serve the public interest.

In this piece, we elaborate on our opinion that a professional ethical guideline for marine conservation communication is necessary. We also report on discussions from a focus group titled, "Overcoming ethical challenges in marine conservation communication" held at the 5th International Marine Conservation Congress (IMCC5). Sixteen marine conservation professionals (scientists, practitioners, and communicators) shared their perspective about existing relationships and modes of engagement between media, journalists and conservation groups, urgency of factual and accurate narratives in ocean conservation, prerequisites of independent and transparent reporting while promoting conservation efforts, and the inclusion of local and indigenous voices in conservation narratives. Focus group participants discussed solutions-driven directives that could be incorporated into a professional code of conduct for conservation communicators and debated the fundamental premises of such a code.

"Freedom of the press is guaranteed only to those who own one" (Liebling, 1964). With the explosion of publishing platforms made possible by the Internet, this quote from Liebling rings truer today than it did in the 1960s. The demise of the journalistic watchdog and the rise of the citizen journalists (Bruns, 2008) have created a dynamic that means it is up to the reader to navigate between professional journalism, irresponsible click-bait, opinion blog posts, and agenda driven articles. Grassroots reporting (blog indexes, personal blogs) and the rise of citizen journalism have created an active audience that not only follows the news, but contributes (Bruns, 2008). The journalistic role of gatekeeping, filtering information before publishing, has diminished, transforming the role instead to gate watcher (Bruns, 2008) or scout in the jungle of information (Brüggemann, 2017), leaving journalists to filter information which is already published. With no dedicated watchdog, open publishing platforms allow everyone with access to the internet to have a voice. This, in turn, is enabling content that is directly or indirectly guided or influenced by those who may carry subjective, agenda-driven intentions, be it an organization, NGO, advertiser, broadcaster, or individual science communicator.

This is blurring the boundaries between environmental journalism and advocacy (Rosenstiel et al., 2016) which we, the authors, believe can have both a positive and negative impact on the way readers understand and interpret marine conservation. In some fields this is allowing more

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****Correspondence:***
Lucy E. Erickson
l.erickson.e@gmail.com

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TABLE 1 | Suggestions to include in a code of professional ethics for marine conservation communicators.

Suggested guideline	Reference or origin
(A) TRANSPARENCY	
(i) Journalistic works must always be independent, and should not be influenced by funding or other benefits, gifts, favors	Adapted from the Professional Journalists (SPJ) Code of Ethics (2014)
(ii) Commissioned articles, or those in return for free travel, favors, extended reach, etc., (from influencers, bloggers, writers) should clearly state that they are sponsored content	Federal Trade Commission's Endorsement Guides, 2017
(iii) Ensure correct attribution of content and images. Respect the rules of quoting	SPJ
(B) BOUNDARIES	
(i) Take a proactive role in enabling journalists to maintain its editorial independence while working as a partner in conservation	IMCC5 Focus Group Discussions
(ii) Accept that you may not have access to the final copy of a journalistic article before it is published, but offer to review/fact-check sections that may be complex	Adapted from Borel et al. (2018)
(iii) Accept that your activities and projects are subject to public scrutiny and accountability through the mainstream media	IMCC5
(C) ENABLING SCIENTIFIC LITERACY	
(i) Support scientists and project leaders to engage directly with media, provide media training	IMCC5; Ocean Media Institute, 2019; Stempra guide to being a media officer, 2019
(ii) Provide access to information by publishing research in open access journals if possible or make clear that you can share copies of papers	IMCC5
(iii) Suggest other reliable resources where possible. Provide outlets for further action or more information	IMCC5
(iv) Take responsibility for the accuracy of the work	Adapted from SPJ
(v) Explore and engage other disciplines in your work to create a richer, more inclusive story e.g., marine science, social science, history	IMCC5; Savoie, 2017
(vi) Craft a story of the science that engages our humanity. In order to be effective, the science narrative can no longer simply inform; it must engage the public by incorporating human agency into the story	Savoie, 2017
(D) ACCURACY AND HONESTY	
(i) Ask yourself if you have exaggerated the significance of your work/findings or if there are other possible interpretations of results	Stempra, 2019
(ii) You may need to use an attention grabbing headline, but commit to including the nuance and context and reality in the rest of the article	Stempra, 2019
(iii) Ask yourself if your personal beliefs have influenced your interpretation of the science	Authors
(iv) Communicate negative as well as positive impacts	ISEAL
(v) Research that has not been peer-reviewed, replicated, or carefully vetted should not be the primary basis of content	IMCC5
(vi) Embrace an approach to science communication that is genuinely evidence-based to minimize polarization	Kahan, 2014
(vii) Think carefully about the use of visual representations so that they convey the meaning you intend	Authors
(viii) Acknowledge the technological limitations in different parts of the world and consider how it may impact your ability to follow-up and fully communicate a story	Authors
(E) DIVERSITY AND INCLUSION	
(i) A local community must be more than a story or data provider to a foreign NGO. Provide a platform for the underrepresented, yet critical voice. Include local and/or indigenous perspectives whenever possible	IMCC5; Aini and West, 2018
(ii) Approach scientific knowledge and indigenous knowledge not as mutually exclusive competing ideas, but as complimentary	Authors
(iii) Embrace diversity (racial, sexual, cultural, gender, age) in storytelling	Adapted from SPJ
(F) INSPIRING ACTION	
(i) Doom and gloom does little to motivate audiences to fully invest in an issue. Point audience toward models of hope and success, even when communicating negative results	Balmford and Knowlton, 2017
(ii) Where appropriate, pair the conservation issue with positive solutions-driven action and/or a call to action	Dyer, 2015

extensive reporting on events such as climate conferences (Rosenstiel et al., 2016), yet it is also creating the opportunity for self-promotion, which depending on the agenda of the writer, can pose threats to the public's objective understanding of marine conservation and the issues facing the planet.

Recently, an opinion piece published in the New York Times discussed the trend of "just add water" that is seeing the triumphant announcement of large marine protected areas, which are protecting relatively empty waters as opposed to prioritizing coastal habitats that are home to 25% of all marine species (Rocha, 2018). While this opinion is not shared by all scientists (MacPherson, 2018), others (Barnes et al., 2018) stress the need to report outcomes as opposed to area when it comes to announcing new protected areas, arguing that the focus

should be on anticipated biodiversity gains rather than the square kilometers protected. The root of this problem may, in fact, lie with the issuer of the original press release (e.g., an NGO) or the opinion piece may have been politically motivated. Crucially, there is an issue of transparency which needs to be improved from both sides: the source (the issuer of the press release) and the entity covering the story.

Another difficulty is simply distinguishing between science journalism—a responsibility to inform and educate the public (Xu, 2013), assess, critique and contextualize science and scientists rather than promote it (Borel, 2015), and science communication, which explains how a natural phenomenon works or describes "the how" of new scientific discoveries. The important difference is that how science communicators portray their topic depends on their intentions, which should be transparent (Borel, 2015). Xu argues that the ability for scientists to publish directly to the general public, without going through the official publishing process, is creating a new "science-media ecosystem." Though a direct link between scientists and the public can be beneficial, journalist Brooke Borel highlights the importance of journalistic scrutiny in questioning the intentions of scientists or organizations publishing their own scientific outreach (Borel, 2015).

While some argue that we are in an "Unlikely Golden Age" at the height of production in terms of both quantity and quality of science and environmental journalism (Hayden and Check Hayden, 2018), we believe that despite this, there is a lack of capacity when it comes to reporting on marine conservation. An example is a recent article by The Guardian (Summers, 2018) which reports on a new scientific study concerning a controversial whale shark tourism site in the Philippines (which one of the authors and her team studies). The journalist reports only one side of the controversy, omitting all previous research from the same study site, and fails to include an outside quote. The article also reported illegal activity which lacked original sources.

The implications of poorly executed journalism such as this are far reaching. They can miseducate the public on complex topics and undermine conservation efforts. Are journalists at capacity and not able to dedicate their full time to covering marine conservation, similar to that of other environmental journalists (Detjen et al., 2000)? Or are there simply too few experts, with only a small group of journalists producing the vast majority of coverage (Brüggemann and Engesser, 2014)? Either way, how can this knowledge gap be moderated?

Communication professionals in NGOs can mitigate a lack of journalistic capacity in the marine conservation space if they commit to balanced, transparent self-reporting, and to help independent and objective reporting. However, this is not always the case. Mongabay's 2016 *Conservation, Divided* series highlighted that the biggest NGOs often issue "press releases that could convince a misanthrope to love people [and] make whatever they do sound like a resounding success, even when the reality is much more complex" (Hance, 2016). Biased self-reporting can also be off-putting for donors. A recent analysis commissioned by ISEAL, the global best practice community of standard setters, found that funders are more likely to believe communications that contain negative as well as positive impact (Chilvers, 2017). This suggests there is an opportunity for professional marine conservation communicators to contribute to objective reporting while improving relationships with key partners.

Finally, whether or not coverage of marine conservation efforts is the result of sponsored or embedded arrangements (e.g., a journalist given access to a remote marine location through an NGO sponsorship), conservation communicators must permit journalistic contents to be produced independently with objectivity and independence needed in the persuasion of the truth.

We believe there is a need for a code of professional ethics for marine conservation communicators that promotes trust, accountability, independence, and solutions-based reporting, all the while furthering the value of a compelling story. These guidelines (**Table 1**) draw together existing resources as well as emerging areas of focus and can be used as a tool by communication practitioners and scientists to create a professional ethics code, but can also be adopted by journalists and other content creators. Acknowledging the limitations of our own knowledge and the small sample size of opinions collated, we present them not as a final or comprehensive list, but as a starting point for much needed future collaborative work in this space.

In conclusion, no matter how much conservation groups have sway over social media and public relation platforms, those are not a replacement for independent journalism. While we should continue to strongly advocate for conservation, we should not make it difficult for journalists to inform the debate with facts and a commitment to making all voices heard. We hope this article will spark a conversation about the necessity of a code of professional ethics for marine conservation communicators.

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MU conceived the idea and scope of problem. GS, LE, MU, and SS devised main conceptual ideas and designed the article. SS and LE took notes at the focus group discussion. All authors provided critical feedback and contributed to develop the final manuscript.

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Shelving the Coast With Vertipools: Retrofitting Artificial Rock Pools on Coastal Structures as Mitigation for Coastal Squeeze

Alice E. Hall[1], Roger J. H. Herbert[1], J. Robert Britton[1], Ian M. Boyd[2] and Nigel C. George[2]*

[1] Faculty of Science and Technology, Life and Environmental Sciences, Bournemouth University, Poole, United Kingdom, [2] Artecology Ltd., Sandown, United Kingdom

Coastal squeeze caused by sea level rise threatens the size, type, and quality of intertidal habitats. Along coastlines protected by hard defenses, there is a risk that natural rocky shore habitats will be lost, with the remaining assemblages, characteristic of hard substrata, confined to sea walls and breakwaters. These assemblages are likely to be less diverse and different to those found on natural shores, as these structures lack features that provide moist refugia required by many organisms at low tide, such as pools and crevices. Engineering solutions can help mitigate the impact of sea level rise by creating habitats that retain water on existing structures. However, as experimental trials are strongly affected by local conditions and motivations, the development of new techniques and solutions are important to meet the needs of local communities and developers. Following a small-scale community project, a feasibility study retrofitted five concrete-cast artificial rock pools ("Vertipools") on a vertical seawall on the south coast of England. After 5 years, the artificial pools increased the species richness of the sea wall and attracted mobile fauna previously absent, including fish and crabs. The Vertipools had assemblages which supported several functional groups including predators and grazers. Although disturbance of algal assemblages on the seawall from the retrofitting process was still evident after 3 years, succession to full canopy cover was underway. Collaboration between policy makers, ecologists, children and artists produced an ecologically sensitive design that delivered substantial benefits for biodiversity, which can be adapted and scaled-up to both mitigate habitat loss and enhance coastal recreational amenity.

Keywords: ecological enhancement, climate change, sea level rise, seawall, habitat creation, ocean sprawl, ecological engineering

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****Correspondence:***
Alice E. Hall
ahall@bournemouth.ac.uk

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INTRODUCTION

Climate change and biodiversity loss are two of the greatest threats to coastal areas around the world (IUCN, 2016; Committee on Climate Change UK, 2018; WWF, 2018). Not only do coastal areas support important biodiversity, act as carbon sinks and provide a buffer in front of sea defenses, they also have an important role in tourism and the provision of cultural services (Committee on Climate Change UK, 2018).

Coastal protection provided by seawalls forms a barrier between the land and sea, preventing the natural migration of the coastline. These barriers, coupled with increased sea levels, are resulting in "coastal squeeze," which occurs when the high water mark is fixed by a defense structure and the low water mark is moving landward due to sea level rise, resulting in substantial losses of intertidal habitats (Pontee, 2011). The construction of coastal defense structures results in the steepening of the shore profile (Jackson and McIlvenny, 2011; Committee on Climate Change UK, 2018) which creates less space for colonization and compressed species zonation (Kendall et al., 2004). This, coupled with the lack of water retention which would naturally occur in crevices and pools (Firth et al., 2013), results in a poor quality habitat. Habitat heterogeneity is also generally absent on most artificial structures; in contrast, natural rocky shores have a high variety of surface textures, crevices, overhangs and pools, which provide suitable refugia and habitats for a diverse range of species (Connell, 1972; Underwood et al., 2008).

Water retention is important on a rocky shore as it creates refugia from desiccation stress and predation during periods of low tide (Firth et al., 2013; White et al., 2014). Although the physico-chemical composition of rock pools is known to fluctuate diurnally and seasonally with changes in temperature, pH, salinity and oxygen saturation, these fluctuations are not as extreme as on the emergent rock surfaces (Daniel and Boyden, 1975; Metaxas and Scheibling, 1993). Rock pools are known to extend the limits of distribution for intertidal species, including larger brown algae (*Fucus* spp.), limpets and mussels (Green, 1971). Photosynthesis of algae within the pools can influence oxygen and carbon dioxide levels, which in turn affect the pH of the water (Metaxas and Scheibling, 1993; Björk et al., 2004). Intertidal fish also use rock pools as habitats, but rock pools decline in abundance as the shore height increases (Bennett and Griffiths, 1984; Zander et al., 1999). White et al. (2014) found that more complex rock pools with ledges and algal cover resulted in higher abundances and diversity of intertidal fish.

Ecological enhancement schemes that integrate ecology with engineering can create potential solutions to mitigate low habitat heterogeneity and produce multifunctional structures that provide coastal protection and also incorporate suitable habitats for marine species (Firth et al., 2013; Dafforn et al., 2015; Evans et al., 2015; Morris et al., 2016). Trials have indicated that creating artificial features which increase water retention and habitat heterogeneity can provide opportunities for colonization by a variety of species (Chapman and Blockley, 2009; Browne and Chapman, 2011; Firth et al., 2014a; Evans et al., 2015). There have been several examples of trials including the use of artificial panels (Moschella et al., 2005; Borsje et al., 2011; Loke and Todd, 2016), manipulations of concrete (Chapman and Underwood, 2011; Firth et al., 2016a), drilling holes into existing artificial structures (Martins et al., 2010; Firth et al., 2014b; Evans et al., 2015; Hall et al., 2018) and by creating pre-cast concrete units [See Firth et al. (2016b) for review]. Browne and Chapman (2011, 2014) deployed pre-cast flowerpots at different tidal heights on a seawall in Sydney to mimic natural rock pools; although some flowerpots were dislodged by waves, the remaining ones increased the biodiversity of the seawall by attracting novel sessile species. Morris et al. (2017) investigated the larger scale effects of the flowerpots by studying the mobile communities. In addition, they added artificial turf to some of the pots in order to see the effects on the native and non-native sessile communities. The study found higher densities of mobile and sessile species in pots without artificial turf, implying that the turf prevented particular species from colonizing the pots. The outcome of experimental trials are strongly affected by local conditions and motivations (Airoldi et al., 2005), therefore development of new techniques and solutions are important to meet the needs of local communities and developers.

Shelving the Coast Project

As part of the community science project "*Shelving the Coast*" on the Isle of Wight in southern England, artists, school children and ecologists designed and created a series of structures subsequently named and referred to here as "Vertipools." These pre-cast concrete artificial rock pools aimed to vertically extend the intertidal zone to mitigate the effects of sea level rise, which could result in the loss of the existing intertidal rock pools. The Vertipools were designed to be attached to coastal structures, such as vertical seawalls and groynes. The exterior patterns were designed by local primary school children aged between 5 and 7, as part of an educational project on coastal squeeze. The V-shape was inspired by the bow of a ship to deflect wave energy and the hollow insides allow for water retention at low tide (**Supplementary Figures S1A–F**). The educational program involved three activities; an animation project, song writing and recording exercise and designing the Vertipools exterior surfaces.

The main purpose of this study was to monitor the growth of the assemblages on and within the Vertipools and to determine whether the installation would improve the species diversity of assemblages living on the seawall within the study area. Studies on colonization of hard substrata have shown changes in assemblages and successional processes with an increased number of species and functional groups over time (Benedetti-Cecchi, 2000; Jenkins and Martins, 2010; Herbert et al., 2017). Our general hypothesis is that there will be a community succession over time in the artificial pools and the disturbed areas of the sea wall. The Vertipools were surveyed seasonally in year 2 and 3 to determine the early community succession and then again in year 5 to compare the species richness with the seawall. The following hypotheses were tested:

(1) Vertipools will support a higher species richness than the seawall after 5 years;
(2) Season and year will have a significant effect on the community succession and in turn the species richness and functional groups of the assemblages within the Vertipools during 2015 and 2016;

In order to assess the impact of Vertipool construction on existing assemblages on the seawall the following hypothesis was tested:

(3) The disturbed area of the seawall will have a similar percentage cover of algae to the undisturbed seawall after 3 years.

MATERIALS AND METHODS

Study Site

The site at Bouldnor is located on the south coast of England approximately 5 km east of Yarmouth on the north west coast of the Isle of Wight (50°42′27.5″N 1°28′57.1″W) (**Figure 1**). The shore is moderately sheltered with a north facing aspect and a mean tidal range of 2 m. This stretch of coast has been heavily modified and includes a vertical concrete seawall constructed in 1985 (**Figure 1**). The seawall has well-established zones of marine algae dominated by the brown seaweeds *Ascophyllum nodosum* and *Fucus spiralis*. Below the sea wall, the shore is dynamic and truncated, consisting of limestone boulders surrounded by mobile mixed sediments. Natural rock pools which are approximately 30 cm in diameter and 10 cm deep, appear when beach levels are low but are often smothered by sediment. Other natural rock pools occur on nearby protected areas. This site was chosen for the trial due to the low risk of public interference and ease of access for installation and monitoring.

Vertipool Description

During September 2013, five concrete wooden-cast Vertipools were installed between Mean Tide Level (MTL) and High Water Neaps (HWN) on the vertical concrete seawall. The outside

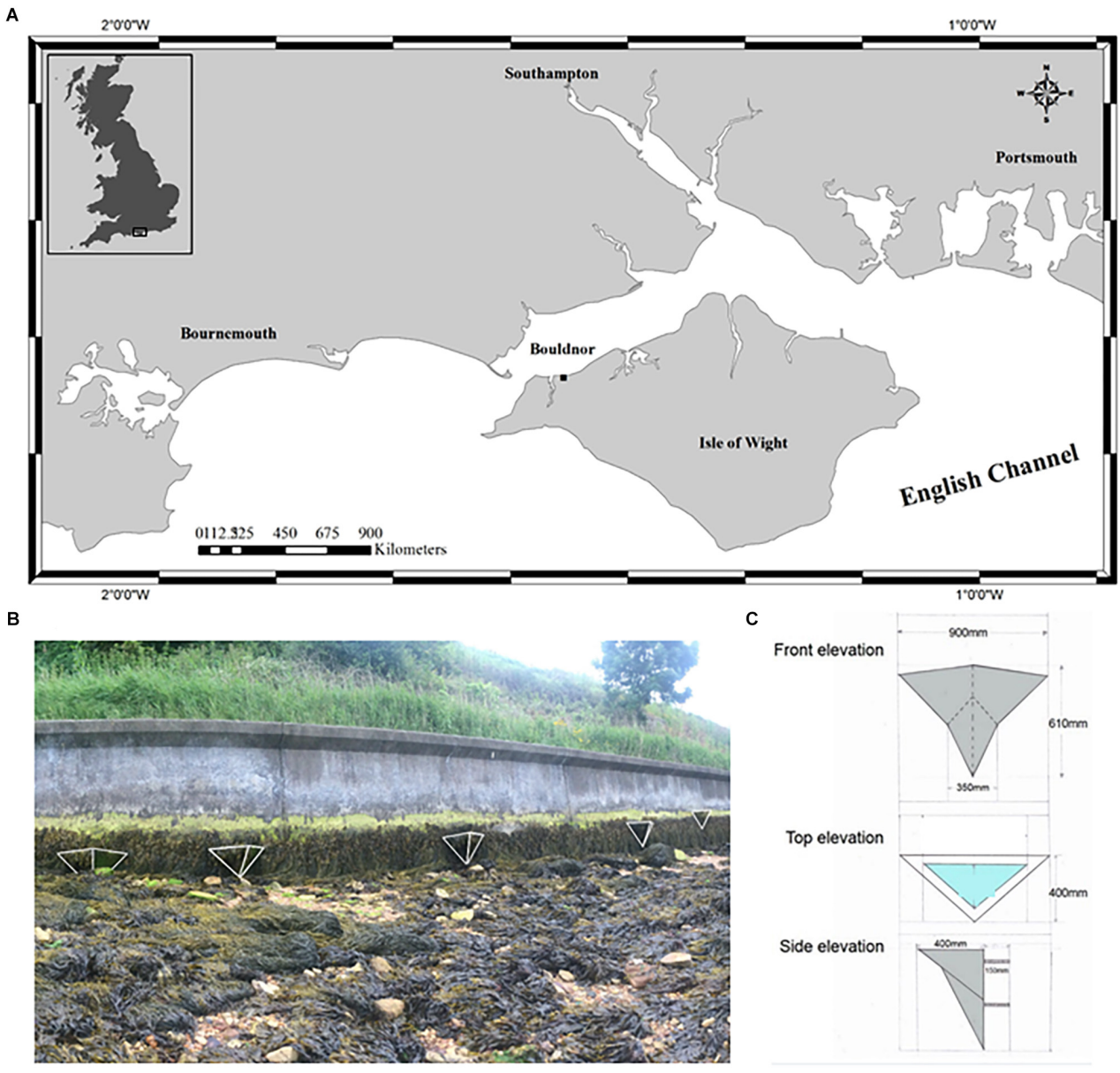


FIGURE 1 | (A) Map illustrating the location of Vertipool test site at Bouldnor, Isle of Wight, **(B)** Location of the seawall and Vertipools on the seawall, and **(C)** dimensions of the Vertipools.

of each Vertipool was hand sculptured using wet cement to incorporate the design of the school children (**Supplementary Figure S1A**). The Vertipools weigh 50 to 70 kg and are 900 mm at their widest, 610 mm in height, protrude a maximum 400 mm from the seawall and have an undulating pool depth of 10–200 mm (**Figure 1** and **Supplementary Figure S1**). Three 20 mm diameter M20 stainless steel coach bars attached to a T-shape steel plate were cast into the back of the Vertipools for attachment to the seawall. Three holes were made in the seawall using a 25 mm diameter SDS drill bit to a depth of 150 mm and filled with a marine grade resin bonding agent (Fischer Resin Mortar), before attaching the Vertipools.

Assemblage Monitoring

Seasonal variation in assemblages in the Vertipools was recorded in spring, summer, autumn and winter of the second and third year after installation (2015 and 2016) and long-term changes were recorded in the fifth year after installation (2018). In addition, the effects of retrofitting on the seawall were monitored in the third year after installation (2016), after which the beach level had risen too much to access some of the control quadrats, so sampling was stopped. All fauna and macroalgae were surveyed thoroughly in each habitat using visual *in situ* non-destructive sampling techniques and organisms were identified to the lowest taxonomic level possible

TABLE 1 | Mean abundance of all species found in the four habitats studied; VP Inside (2015, 2016, 2018 *n* = 50), VP Outside (2015, 2016, 2018 *n* = 50), Seawall (2016, 2018 *n* = 20), Disturbed Seawall (2016 *n* = 10).

	Mean abundance			
Species	**VP inside**	**VP outside**	**Seawall**	**Disturbed Seawall**
Phylum Cnidaria				
Actinia equina (c)	0.02	0.00	0.00	0.00
Phylum Annelida				
Spirorbis spirorbis (c)	2.37	0.12	0.65	0.30
Sub Plylum Crustacea				
Austrominius modestus (c)	0.00	0.10	0.20	0.00
Carcinus maenas (c)	0.13	0.02	0.00	0.00
Idotea granulosa (c)	0.01	0.00	0.00	0.00
Ligia oceanica (c)	0.00	0.02	0.00	0.00
Porcellana platycheles (c)	0.04	0.00	0.00	0.00
Semibalanus balanoides (c)	0.00	0.44	2.25	0.00
Phylum Mollusca				
Gibbula umbilicalis (c)	0.02	0.00	0.00	0.00
Littorina littorea (c)	0.01	0.00	0.00	0.00
Littorina obtusata (c)	0.07	0.14	0.25	0.30
Patella vulgata (c)	0.09	0.06	0.10	0.00
Phylum Chordata				
Lipophrys pholis (c)	0.01	0.00	0.00	0.00
Phylum Chlorophyta				
Chaetomorpha sp. *(%)*	15.11	0.00	0.00	0.00
Cladophora rupestris (%)	6.89	2.40	31.40	69.00
Ulva linza (%)	6.81	1.40	8.80	0.00
Phylum Rhodophyta				
Catenella sp. *(%)*	0.07	2.80	2.80	3.80
Ceramium sp. *(%)*	1.16	0.00	0.00	0.00
Plocamium sp. *(%)*	0.02	0.00	0.00	0.00
Polysiphonia sp. *(%)*	0.88	0.00	0.00	0.00
Porphyra sp. *(%)*	0.00	0.22	0.00	0.00
Rhodothamniella floridula (%)	0.07	6.30	54.35	26.50
Phylum Ochrophyta				
Ascophyllum nodosum (%)	0.05	6.70	68.20	2.60
Ectocarpus sp. *(%)*	12.03	0.00	0.00	0.00
Fucus serratus (%)	2.28	0.14	2.75	7.00
Fucus spiralis (%)	30.76	7.80	17.90	19.00
Halurus sp. *(%)*	0.08	0.00	0.00	0.00
Sargassum muticum (%)	0.82	0.00	0.00	0.00
Total No. species	**24**	**15**	**12**	**8**

c, count data; %, percentage cover.

(Evans et al., 2015). Care was taken to survey both the canopy and understory communities.

The Vertipools were sampled seasonally (spring, summer, autumn, and winter) during 2015 and 2016 in order to detect any variation in assemblages. Percentage cover of macroalgae and counts of sessile and mobile fauna were recorded on both the inside and outside of each Vertipool separately. The internal and external surface area of the pool were each approximately 0.25 m^2. In addition, 5 years post installation, the Vertipools were surveyed again once in September 2018 and the communities were compared to the adjacent seawall.

The undisturbed seawall adjacent to the Vertipools was sampled using five 0.25 m^2 quadrats to record the percentage cover of algae and counts of sessile and mobile fauna at a comparable tidal height (>2 m distant) adjacent to the Vertipools. This data was compared with the data collected in the Vertipools after 5 years.

To measure the effect of retrofitting the Vertipools on the existing algal assemblage on the vertical sea wall, areas scraped and cleared at the time of installation (referred to as "Disturbed Seawall") were monitored in 2016 (3 years post installation). The limited space available necessitated use of smaller 25 cm × 25 cm quadrats placed either side of each of the five Vertipools to record percentage cover of flora (N = 10).

Statistical Analyses

All data were tested for normality and equal variances and all t-tests and Generalized Linear Mixed Models (GLMMs) were run using R v3.5.1 (R Core Team, 2018), the GLMMs were run using package "lme4" version 1.1-18-1. Species richness (S) were calculated for each Vertipool using the DIVERSE function, which calculates diversity indices, in PRIMER-e V6 (Clarke and Gorley, 2006). To test hypothesis 1, that Vertipools will support greater species richness than the seawall after 5 years, the data recorded from the inside and outside habitats on each Vertipool were combined and averaged (N = 5). A two-sample t-test was used to test for differences in mean species richness between habitats (Vertipool and control seawall).

To test hypothesis 2, whether species richness and abundance of algae in the Vertipools were significantly different between seasons (spring, summer, autumn, and winter) and years (2015 and 2016), a Poisson generalized linear mixed model was used (R Core Team, 2018). The Poisson GLMM included the individual Vertipool as a random variable within the model. The parameters were fitted by maximum likelihood (Bolker et al., 2009).

To test if seasonal variation in assemblages and functional groups were observed in Vertipools during 2015 and 2016, PRIMER-E was used on presence/absence data to create a Jaccard similarity matrix. Assemblages were presented visually using Multidimensional Scaling (MDS) and ANOSIM (analysis of similarity) was used to test for differences in assemblages between seasons and years. All species were classified into the morph-functional groups that were present at the site: Canopy algae, Sub-canopy algae, Filter feeders, Grazers, and Predators (Arenas et al., 2006; Firth et al., 2014a). The five functional groups were used to create a Jaccard similarity matrix on presence/absence data. ANOSIM was used to highlight the variation in functional groups between Season and Year and significant results were explored further using SIMPER (similarity of percentages).

To test hypothesis 3, that the disturbed area of the seawall had similar percentage cover of algae to the undisturbed seawall after 3 years, a beta regression GLM ("betareg" version 3.1–2) was used to test the differences in mean percentage cover of algae between habitats (Disturbed seawall and Control seawall). To assess the similarity of communities, a Jaccard similarity matrix on presence/absence data was used perform an ANOSIM and SIMPER to highlight the variation in species abundance.

RESULTS

Over the 5 years, a total of 24 species were recorded on the inside of the Vertipools, 15 species were found on the outside of the Vertipools, 12 species on the control seawall and eight species on the disturbed seawall (**Table 1**). During the study, the algal cover on the exterior of the Vertipools developed from a community dominated by opportunistic green algae (*Ulva* spp.) into a dense over-hanging fucoid canopy (*F. spiralis*). The algal assemblages inside the Vertipools comprised of fucoids, filamentous and branching algae. Mobile species observed inside the Vertipools included fish (*Lipophrys pholis*), crabs (*Carcinus maenas*, *Porcellana platycheles*) and gastropods *Patella vulgata* and *Littorina obtusata*. The shore crab *C. maenas* was observed inside the Vertipools at various life stages including juvenile, adult and when freshly molted along with the discarded exoskeleton indicating recent ecdysis. The broad clawed porcelain crab (*P. platycheles*) was recorded in the Vertipool located at the highest tidal height, a location in which it would previously not have been able to survive without the refuge provided by the Vertipool. The brown algae *Sargassum muticum* found inside the pools and the barnacle *Austrominius modestus* on the exterior surfaces were the only two non-native species observed.

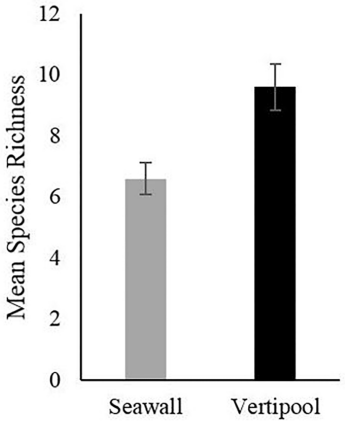


FIGURE 2 | Mean species richness after 5 years recorded in Vertipool and Seawall (Control) (+/– SE., N = 5).

TABLE 2 | Generalized Linear mixed effect model results for Species Richness (S) (all fauna and flora) and Total abundance of algae (N) between season, year and season×year with Vertipool included as a random effect for 2015 and 2016 (***$P < 0.001$, **$P < 0.01$,*$P < 0.05$, NS, Not significant).

(a)	Species richness (S)			Total abundance of algae (N)		
	df	***F*-value**	***P*-value**	**df**	***F*-value**	***P*-value**
Fixed effects						
Season	3	0.359	0.031*	3	3.533	0.005**
Year	1	4.163	0.342 NS	1	6.693	0.008**
Season × Year	3	1.8720	1.000 NS	3	2.576	1.000 NS
Random effects	**df**	**Variance**	**Std. Dev**	**df**	**Variance**	**Std. Dev**
Vertipool	5	0.039	0.199	5	51.84	7.20

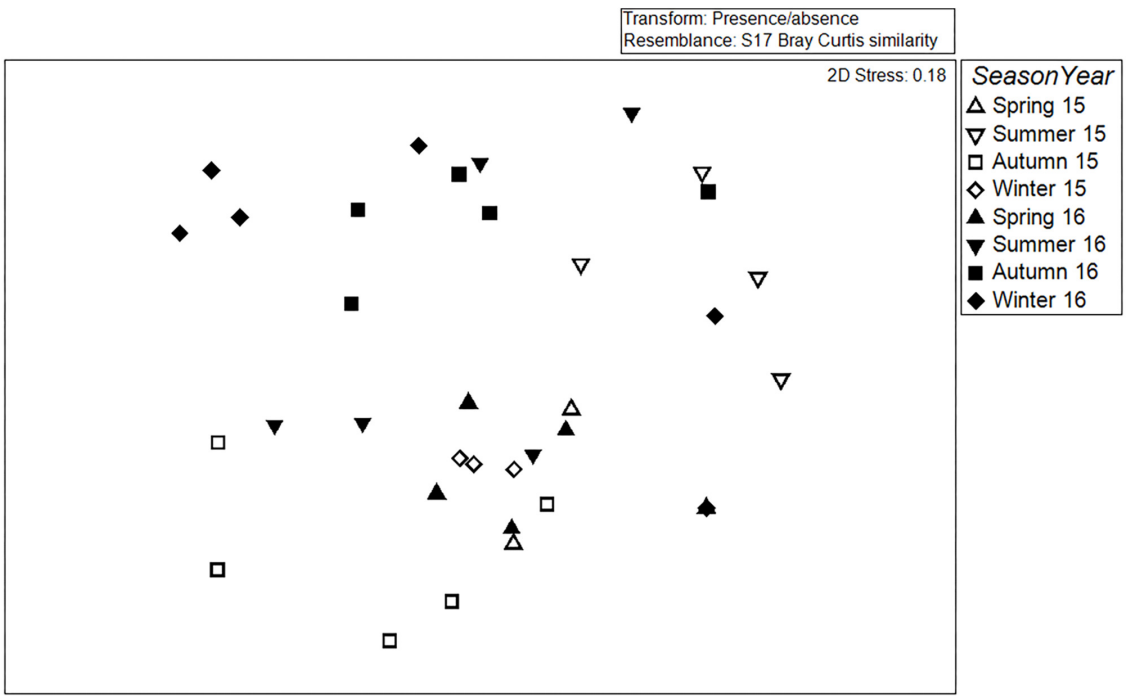


FIGURE 3 | Multidimensional Scaling (MDS) plot comparing assemblages inside the Vertipools in Spring 2015–Winter 2016.

Vertipool vs. Seawall

After 5 years the Vertipools showed a significantly greater species richness (mean = 9.6) compared to the seawall (mean = 6.6) (t-test$_8 = -3.32$, $P = 0.01$) (**Figure 2**). Ten species were found to be unique to the Vertipools; these were algae *Chaetomorpha* sp. *Ulva lactuca, Ectocarpus* sp., *F. serratus* and *Polysiphonia* sp., benthic invertebrates *Actinia equina* and *A. modestus* and mobile species *L. obtusata, C. maenas*, and *P. platycheles* (**Table 1**). Additionally, the shanny fish *L. pholis* was found in the pools in March 2015. The barnacle *A. modestus* was recorded on the exterior rough surface of the Vertipool, but not found on the sea wall. Fronds of the alga *F. spiralis* attained a greater tidal elevation on the Vertipool compared to the sea wall (**Supplementary Figure S1F**).

Vertipools

There was a significant difference in species richness between seasons (**Table 2**) with winter having the highest mean species richness and spring having the lowest mean species richness. There was no significant difference in species richness between years and no significant interaction between seasons and year (**Table 2**). There was a significant difference in total abundance of algae between both season and year (**Table 2**), with 2016 having a higher abundance of algae than 2015 and spring having the lowest abundance of algae across all seasons (**Table 2**).

There was a significant difference in assemblage composition among seasons (Global $R = 0.143$, $P = 0.0002$) and between years (Global $R = 0.117$, $P = 0.001$, **Figure 3**). The MDS showed that the assemblages in 2015 and 2016 are separated with slight overlap and the spring samples are clustered together (**Figure 3**).

The functional groups of species inside the Vertipools were also significantly different between years (ANOSIM, Global $R = 0.083$, $P = 0.026$), but not seasons (ANOSIM, Global $R = 0.012$, $P = 0.336$, **Figure 4**). The Vertipools were initially dominated by sub-canopy algae, yet after spring 2016 it was

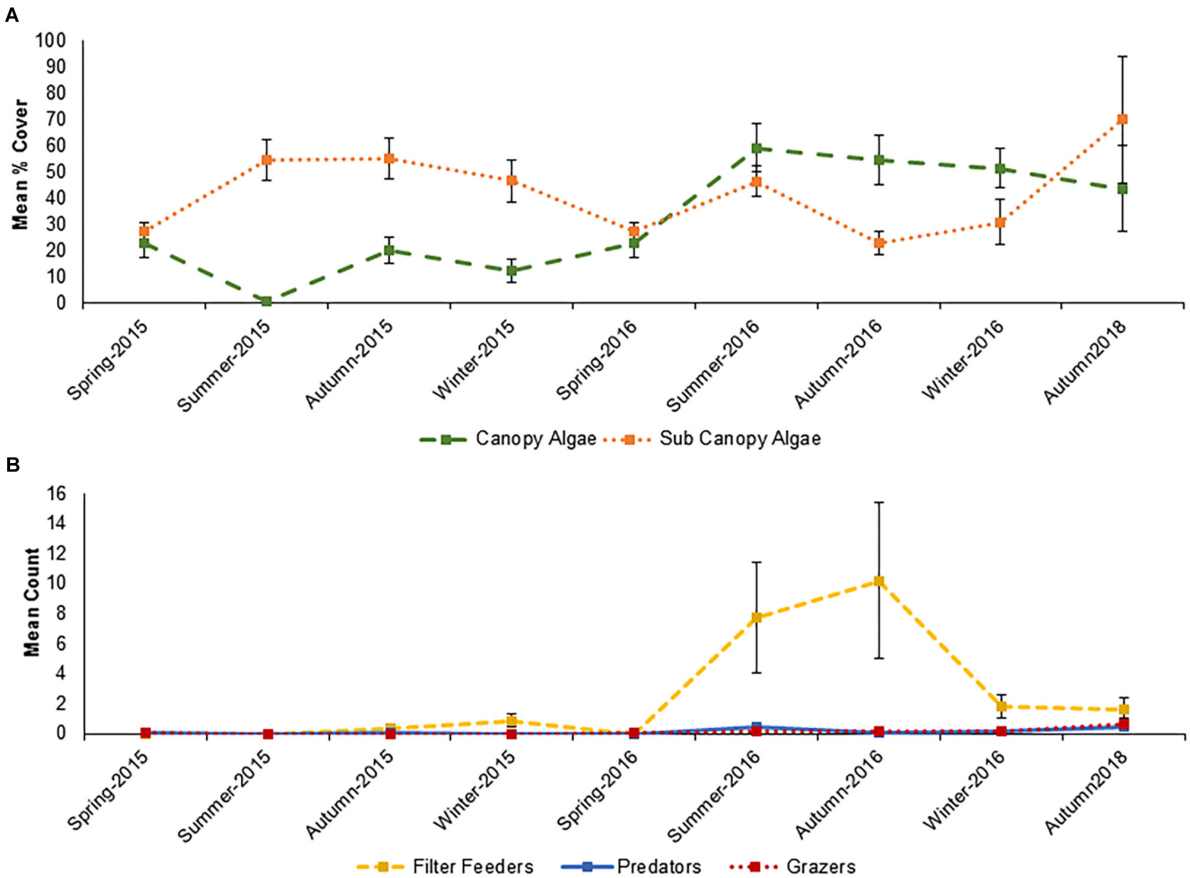


FIGURE 4 | Seasonal variation in **(A)** mean % cover and **(B)** count of main functional groups inside the Vertipools between Spring 2015 and Autumn 2018 (+/–SE., *N* = 5). The rock pools were fitted to the wall in September 2013.

TABLE 3 | SIMPER analysis run using Jaccard similarity matrix indicating average abundance of functional groups per Vertipool in 2015 and 2016 (Av. Abund = mean abundance, Av. Diss = Average dissimilarity, Diss/SD = SD of dissimilarity, Contrib% = contribution %, Cum% = cumulative percentage).

Functional group	2015 Av. Abund	2016 Av. Abund	Av. Diss	Diss/SD	Contrib.%	Cum.%
Canopy algae	13.97	47.02	9.04	0.7	30.83	30.83
Filter feeders	0.3	4.93	8.73	0.88	29.78	60.61
Predators	0.03	0.15	6.08	0.7	20.75	81.36
Grazers	0.02	0.12	5.47	0.67	18.64	100

transformed into canopy-algae dominated community. Spring 2016 heralded the arrival of filter feeders and predators in the Vertipools (**Figure 4**). **Table 3** details the average abundance of functional groups for each year; canopy algae, filter feeders, predators and grazers were all more abundant in 2016.

Disturbed Seawall vs. Undisturbed Seawall

After 3 years, the disturbed seawall communities had not recovered algal percentage cover similar to that of the undisturbed sea wall (beta regression GLM, $F = 47.87$, $P < 0.001$, pseudo r-squared = 0.66, **Figure 5**). Yet there was a significant difference in assemblage composition between the disturbed and undisturbed seawall (ANOSIM, Global $R = 0.193$, $P = 0.18$). 78.03% of the overall 33.69% dissimilarity between habitats was due to four algal species; *A. nodosum*, *F. spiralis*, *F. serratus*, and *Catenella* sp. *A. nodosum* was most abundant on the seawall, whereas *F. spiralis*, *F. serratus*, and *Catenella* sp. were most abundant on the disturbed seawall (**Table 4**).

DISCUSSION

Although a small-scale study, compared to previous studies (Browne and Chapman, 2014; Evans et al., 2015; Firth et al., 2016a) this currently represents the longest time-series of species richness and assemblage compositions in artificial rock pools. The water retention and increased surface texture provided by

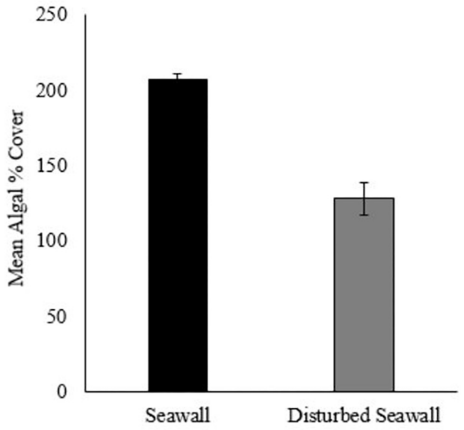


FIGURE 5 | Mean % cover of algae recorded on the disturbed seawall and the control seawall after 3 years (+/– SE, *N* = 10, note difference in quadrat size between habitats- disturbed seawall 25 × 25 cm, control seawall 50 × 50 cm).

the Vertipools created a habitat which was absent from the existing sea wall, enabling a variety of different rock pool species, including fish, to inhabit the structure. Within the study area, the Vertipools increased the species richness on the seawall, supporting a wider range of taxa that is more characteristic of natural shores (Crisp and Southward, 1958; Martins et al., 2007; Firth et al., 2014a). All of the Vertipools remained attached to the seawall with no visible signs of damage; destruction of enhancement devices by waves has been a problem in previous studies (Browne and Chapman, 2014). The Vertipools were designed to deflect wave energy and the strong internal and external fixings ensured that no damage was caused to the Vertipools or the seawall. One of the greatest concerns with retrofitting enhancement devices is the potential to damage the structural integrity of the coastal assets. As sea levels rise and coastal squeeze becomes more severe (Pontee, 2011; Committee on Climate Change UK, 2018), limiting the refugia provided by natural habitats (Jackson and McIlvenny, 2011), it is probable that the Vertipools will become more important to species currently surviving in natural pools at lower tidal levels. Without the refuge provided by the Vertipools, many of the mobile species recorded would not be able to survive at such a high tidal height (Pallas et al., 2006).

Initially, the Vertipools were colonized by opportunistic green algae (*Ulva* spp.), followed by a shift to fucoid algae on the exterior and branching/filamentous algae on the interior. This follows typical succession on a rocky shore (Benedetti-Cecchi and Cinelli, 1996; Benedetti-Cecchi, 2000; Martins et al., 2007; Viejo et al., 2008), although a longer monitoring period is required to establish whether the assemblages have stabilized (Browne and Chapman, 2014). An increased number of functional groups found within the Vertipools in 2016, suggest that communities were still developing; in 2015 the assemblage was dominated by sub-canopy algae, yet in spring 2016 the assemblage changed and became characterized by canopy algae with an increased number of filter feeders, predators and grazers.

Seasonal variation in assemblages has been observed in the Vertipools, with red filamentous algae appearing in the summer months (Christie et al., 2009) and barnacles recruiting in the spring (Jenkins et al., 2000). The close proximity to natural habitat and propagule supply may facilitate colonization at this site and locations with less spatial connectivity may take longer to colonize (Cowen and Sponaugle, 2009; Herbert et al., 2017). The largest change in the community was noted when grazers, particularly limpets, moved onto the Vertipools, removing the fucoids from the exterior of the Vertipools.

The Vertipool located at the greatest height on the seawall took longest to colonize, with the interior community predominately consisting of opportunistic algae (*Ulva* spp.), whereas the exterior was colonized by *F. spiralis*. Previously, the reduced number of organisms in high shore ecological enhancements has been linked to low recruitment levels (Browne and Chapman, 2014). Over the duration of the study it was noticed that the elevation of fucoids on the exterior of the Vertipool increased to a height above that of the fucoids growing on the seawall, possibly due to the damper, shaded "overhang" effect created by the Vertipools.

Enhancement of Seawall

Given predicted sea level rise and truncation of the intertidal zone, the provision of suitable habitats on artificial structures is

TABLE 4 | SIMPER analysis run using Jaccard similarity matrix on disturbed vs. control seawall after 3 years, mean abundance (%) of species contributing to dissimilarity between habitats SIMPER, Average dissimilarity = 39.63% (Av. Abund = mean abundance, Av. Diss = Average dissimilarity, Diss/SD = SD of dissimilarity, Contrib% = contribution %, Cum% = cumulative percentage).

Species	Control seawall Av. Abund	Disturbed seawall Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Ascophyllum nodosum (%)	85.5	2.6	8.78	1.46	26.07	26.07
Fucus spiralis (%)	12	19	6.56	1.03	19.47	45.54
Fucus serratus (%)	2.5	7	5.75	0.86	17.07	62.6
Catenella sp. *(%)*	1.5	3.8	5.2	0.87	15.43	78.03
Spirorbis (c)	1.1	0.3	4.12	0.7	12.23	90.26
Semibalanus balanoides (c)	0.1	0	1.16	0.33	3.44	93.7
Littorina littorea (c)	0	0.3	1.09	0.33	3.22	96.92
Rhodothamniella floridula (%)	60.5	26.5	1.04	0.33	3.08	100

c = count data; % = percentage cover.

necessary to prevent further biodiversity loss. As with natural rock pools found on the upper shore, the distribution of intertidal species extended higher up the shore due to the installation of the Vertipools. Mobile fauna such as crabs (*C. maenas*, *P. platycheles*) and periwinkles (*L. obtusata*), previously absent from the seawall, were found inside the Vertipools on multiple occasions. Studies have shown that rock pools support a more diverse community than adjacent rock faces (Firth et al., 2013, 2014a) on both natural shores (Firth et al., 2014a) and artificial structures (Chapman and Blockley, 2009; Browne and Chapman, 2014; Evans et al., 2015). As this study was a small-scale trial project, the five Vertipools were only installed at one site, resulting in low spatial replication. Future studies will need to include trials at multiple sheltered and exposed sites with increased replication to determine wider scale benefits and impacts. Shore height, pool volume, surface area, depth, shading and drainage are known to impact the physico-chemical composition of rock pools (Daniel and Boyden, 1975; Metaxas and Scheibling, 1993; White et al., 2014), therefore, the installation of artificial pools of different sizes and depths and at a variety of heights would be beneficial. The elevated presence of the alga *F. spiralis* on the exterior base of the Vertipools could indicate reduced desiccation stress for the species at this height. Therefore, compared to the seawall, the establishment of Vertipools, constructed of roughened concrete, creates a more heterogeneous habitat providing water retention and damp refugia. These features enable species to occupy levels that are elevated above current zones on the seashore. Natural rock pools are not permanent features of this particular study site as the shore is mobile and prone to periodic smothering by sediments. However, rock pools harboring protected species at similar tidal levels do occur on the island 5 km to the west, so the installation of artificial pools on sea walls may be regionally beneficial to the conservation of these species should they colonize in the longer term.

Impacts of Retrofitting

In the present study, the shoreline and seawall were dominated by *A. nodosum* which is slow to recover after disturbance events (Jenkins et al., 2004), due to poor growth and recruitment mortality (Stengel and Dring, 1997). This study monitored how the retrofitting process affected the existing algal cover on the seawall. After 3 years the disturbed areas were recolonized by *F. spiralis* (19.00% cover) and *A. nodosum* (2.60% cover), with an understory of *R. floridula* (26.50% cover) and *C. rupestris* sp. (69% cover). However, as shown by Jenkins et al. (2004), *A. nodosum* took longer to grow than *F. spiralis*. Overall, the early recolonization of algae indicates that the retrofitting process is unlikely to have any long-term impact on these assemblages, although *A. nodosum* does need continued monitoring. As the disturbed patches of the sea wall were limited in area, a smaller sized quadrat was necessary for sampling, so comparative estimates of algal cover areas should be considered with caution.

One criticism of retrofitted objects is that they might reduce the structural integrity of the seawall (French, 2001). To date, however, no visible signs of damage or weakening have been observed, although this will continue to be monitored over time. Moreover, cross disciplinary work needs to be conducted between engineers and ecologists to create multifunctional structures for the future (Dafforn et al., 2015; Firth et al., 2016a). Evidence suggests that if ecological enhancement devices are aesthetically pleasing, the general public and coastal managers are more supportive of their use (Morris et al., 2016). Incorporating education and public engagement into habitat creation schemes is an excellent way to connect and educate the general public and school children on important issues such as coastal squeeze and sea level rise. The use of art to activate and engage the students on a complex topic worked well in designing the Vertipools exterior patterns and is recommended for future projects.

CONCLUSION

Extending the intertidal zone vertically by creating bioreceptive artificial rock pools for marine life to inhabit has been successful in this feasibility study. Vertipools could be installed and retrofitted on a variety of different coastal structures including seawalls and groynes to produce biologically favorable built environments. If replicated more widely, these features have the potential to mitigate the impact of coastal squeeze and other physical disturbances that limit the size of the intertidal zone, such as coastal development. Combined with other interventions, such as the creation of holes and grooves to create refugia at different scales, increased habitat heterogeneity on these structures will improve species and functional diversity.

AUTHOR CONTRIBUTIONS

AH, RH, JB, IB, and NG contributed to the conception and design of the study. AH conducted the data collection and statistical analysis. AH wrote the main sections of the manuscript with inputs from RH, IB, NG, and JB. All authors contributed to manuscript revision and read and approved the submitted version.

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SUPPLEMENTARY MATERIAL

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Nearshore Fish Aggregating Devices Show Positive Outcomes for Sustainable Fisheries Development in Timor-Leste

Alexander Tilley[1,2], Shaun P. Wilkinson[3], Jeppe Kolding[4], Juliana López-Angarita[5], Mario Pereira[1] and David J. Mills[2,6]*

[1] WorldFish (Timor-Leste), Dili, Timor-Leste, [2] WorldFish (Malaysia), Penang, Malaysia, [3] School of Biological Sciences, Victoria University of Wellington, Wellington, New Zealand, [4] Department of Biological Sciences, University of Bergen, Bergen, Norway, [5] Talking Oceans Foundation, Bogotá, Colombia, [6] Australian Research Council Centre of Excellence for Coral Reef Studies, James Cook University, Townsville, QLD, Australia

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Simon Nicol, University of Canberra, Australia
Reniel Cabral, University of California, Santa Barbara, United States

***Correspondence:**
Alexander Tilley
alex.tilley@gmail.com

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Capture fisheries in small island developing states (SIDS) have the capacity to increase access to vital micronutrient-rich food to tackle malnutrition, but when fishers are restricted to nearshore habitats by limited capacity (boats, engines, fishing gear), fisheries production can be low. This is the case of coastal Timor-Leste, where some of the world's most diverse coral reefs are juxtaposed with one of the world's most undernourished populations. In these settings, interventions that have successfully improved livelihoods from fishing and reduced threats to biodiversity are rare. Elsewhere in the Pacific, nearshore anchored fish aggregating devices (FADs) have shown success in improving catch rates by making pelagic stocks more accessible to small-scale fishers. Here we test the effects of FADs at increasing capture fish production, by deploying eight experimental FADs at four sites around the country and recording catch and effort data from FAD and non-FAD fishing trips. We assess the effects of FADs on catch rates and catch assemblage and the rate of 100% return on investment (RoI). The average longevity of FADs was 11 months. Results show a significant positive effect of FADs on catch rates when controlling for random site variation, with FADs paying for themselves in ∼5 months or less at three out of four sites. Across all sites and fishing types, 63 species were identified, but FAD catches significantly reduced overall assemblage diversity, with three species (*Sardinella* spp., *Decapterus macarellus*, *Rastrelliger brachysoma*) representing 96% of the catch. Despite the relatively short longevity of FADs deployed in Timor-Leste, the fast RoI seen at most sites indicates that FADs are effective in providing livelihood benefits in certain locations. Catch rates were highest where fishers were specialized, invested in FAD fishing, and formed catch sharing groups with access rights to specific FADs. National level investment into a FAD programme by the government could realistically increase overall fish production in the country, thereby improving availability of micronutrient rich fish to combat malnutrition. A deployment program should be coupled with capacity building around group formation and defining access rights to ensure equitable community benefits.

Keywords: FAD, small-scale fisheries, coral reefs, CPUE, nutrition, food security, biodiversity, small pelagic

INTRODUCTION

People living in small island developing states (SIDS) of Asia-Pacific gain disproportionate livelihood and nutrition benefits from nearshore marine habitats (Connell, 2013). Small-scale fisheries (SSF) in these SIDS predominantly rely on coral reef habitats that are increasingly under threat from diverse drivers (Hughes et al., 2003; Pandolfi, 2003; Bellwood et al., 2004), diminishing their ecological function and potentially affecting the well-being of the millions of people with reef-dependent livelihoods (Teh et al., 2013). Localized fishing and coastal development are pervasive drivers of reef degradation (Mora et al., 2011; Hughes et al., 2017), and with increasing climate variability comes greater vulnerability of fishers (Sainsbury et al., 2018). At current rates of population growth, global food production must increase by 60% by 2050 to feed the world (FAO, 2009) yet in SIDS, alternative sources of food and livelihoods are often constrained by physical geography and natural resources, human and financial resources, infrastructure, and viable markets (Feeny and McGillivray, 2010; Campbell et al., 2016). Less than a third of Pacific countries will be able to meet per capita fish consumption demand in 2030 based on current SSF practices (Bell et al., 2009). As such, projections of the food deficit and the required increases in yield underpin national and regional development strategies, but effective, affordable and scalable governance solutions with minimal environmental degradation, remain elusive.

Anchored, nearshore fish aggregating devices (FADs) are suggested as a practical and efficient means of improving local food security and reducing pressure on reefs by making oceanic fish more available and accessible to artisanal fishers, and thereby providing an immediate improvement to food security (Bell et al., 2009, 2015b; Sharp, 2011). There is convincing evidence that since their adoption in the Pacific in the late 1970s, FADs have in many instances substantially increased SSF catch rates (Matsumoto et al., 1981; Désurmont and Chapman, 2000; Dempster and Taquet, 2004; Sharp, 2011). As such many Pacific nations have integrated the deployment of FADs into their national fisheries action plans and policy (Sharp, 2011; Campbell et al., 2016). Research into financial cost-benefit analysis of FAD deployments, even in the artisanal sector, is mostly restricted to FADs anchored far offshore (>12 nautical miles) targeting tuna, with reported return rates of between 40 and 312% (Sims, 1988; Detolle et al., 1998; Sharp, 2011; Guyader et al., 2013). Far fewer studies evaluate returns on investments of nearshore FADs, within three nautical miles of the coast.

In Pacific SIDS, fishing provides the major source of animal protein (Bell et al., 2009), whereas, in Timor-Leste, the fisheries sector is considered to be functioning well below its potential (Alonso Población, 2013; Mills et al., 2013). Since 2002, an independent, post-conflict Timor-Leste has made rapid development progress but faces significant challenges in alleviating poverty and food insecurity. Ranked 10th on the Global Hunger Index, Timor-Leste has the 2nd highest prevalence of childhood stunting in the world (low height for age), affecting more than 50% of children under five (von Grebmer et al., 2018). Recent consumption data suggest a national average fish consumption of only 6.1 kg per person per annum (AMSAT International, 2011), well below neighboring Indonesia (27 kg), or the global average of 20.5 kg (FAO, 2018). Fisheries do play an important role in nutrition and livelihoods for coastal dwellers in Timor-Leste (Mills et al., 2017), who consume almost three times more fish than the national average (AMSAT International, 2011). Yet, a lack of economic incentives to invest in the sector due to poor infrastructure and low economic returns (Hartmann, 2010) jeopardizes the stated development goal of doubling capture fisheries productivity (an increase of 6,500 t) by 2020 (GOTL, 2011).

Proponents of nearshore FAD deployment programs typically advocate for their use in providing artisanal fishers with access to high productivity, high-value tuna fisheries (Bell et al., 2009, 2018). In Timor-Leste, FADs primarily increase access to stocks of small pelagic fish, promoting the use of a high productivity resource (rapid growth, short lifespans, and high mortality rates) better adapted for supporting sustainable fisheries (Dalzell, 1993). Research deployments of FADs in Timor-Leste only started in 2013. Prior to that, a few communities constructed and deployed traditional FADs. A near-complete absence of current data on fisheries yields and income means that quantitative estimates of the impacts of FAD deployment have previously been impossible for Timor-Leste. The aim of this paper is to explore if a national inshore FAD program can contribute to food and nutrition security in Timor-Leste. We use catch data, vessel tracks and interview data from communities participating in a pilot FAD programme by Timor-Leste's Ministry of Agriculture and Fisheries (MAF) and WorldFish to evaluate: (1) How do FADs affect SSF catch and effort rates and catch assemblages using existing and augmented gears?; (2) What is the economic rate of return on investment of FAD deployments?; and (3) What are the key opportunities and constraints to scaling FAD deployment?

MATERIALS AND METHODS

Study Sites and FAD Deployment

For this study, we utilize fishing data from four communities where eight FADs were deployed by WorldFish and MAF between September 2016 and July 2017 (**Table 1**). The evolved design and deployment methods of the FADs follow those detailed in Mills and Tilley (2017).

The four community sites represent the three ecologically distinct zones of Timor-Leste: Biacou and Vemasse are located on the dry north coast of the mainland with very narrow and steep reef shelf, Adarai on the more exposed south coast with a long gradual continental shelf slope, and Adara on the more sheltered west coast of Atauro Island with a steep reef dropoff to >4000 m depth (**Figure 1**).

Fisheries Characterisation and Catch Documentation

To characterize the fisheries in each site, informal interviews and discussions were conducted in study communities during FAD construction and deployment processes, and by data collectors at landing sites. Information on gear types in use, diel and seasonal

TABLE 1 | Deployment depth and longevity of artisanal nearshore fish aggregating devices at four community fishing sites in Timor-Leste.

Site	Depth (m)	Date deployed	Date lost	Duration (months)	Distance from shore (km)
Adara	170	1-Sep-16	Mar-17	7	0.57
	158	29-Jul-17	Apr-18	11	0.45
Vemasse	340	03-Mar-17	Sep-17	6	2.77
	260	03-Mar-17	Mar-18	12	2.35
	250	03-Mar-17	Mar-18	12	1.85
Adarai	65	27-Apr-17	–	18+	2.66
	65	27-Apr-17	–	18+	3.62
Biacou	100	12-May-17	Apr-18	11	1.96
			Average	**11.9 ± 4.4**	**2.03**

Where no "Date lost" listed, FADs are still in place. Gray shading separates sites.

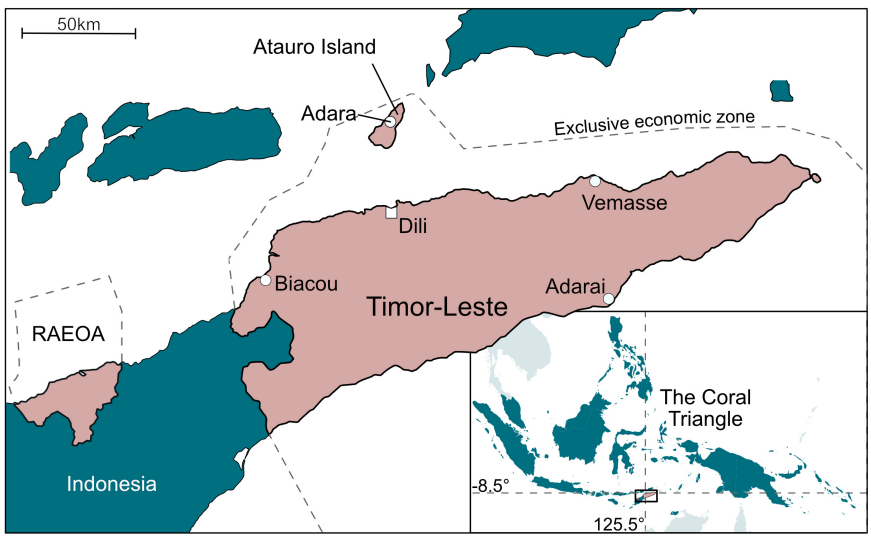


FIGURE 1 | Location of Timor-Leste with the Coral Triangle (inset), and the location of four study sites for testing nearshore FAD deployments. Dotted lines represent the Exclusive Economic Zone of Timor-Leste. The RAEOA is the Special Administrative Region of Oecusse-Ambeno. (Adapted from Tilley et al., 2019. Map created using a vector layer adapted from FreeVectorMaps.com©Striped Candy LLC.)

timing of fishing, fishing frequency, and access and ownership arrangements of fishing grounds and FADs were collected and used to interpret fish landings data.

Landings data were gathered by data collectors in fishing communities between September 2016 and September 2018. To balance the study design and control for year to year variations in abundance and/or catch rates, one year of data between April 2017 and March 2018 was used for analyses of FAD effects on catch rates.

A community member at each of the landing sites was trained in fish identification and the use of a tablet-based survey form developed using the mobile survey software suite KoBoToolbox[1] for Android devices. Data entered on the tablet was uploaded to a central online database using KoboToolbox via a 3G connection. Only the catches of fishers based at each village site were recorded at the landing site associated with that village. We also recorded fishing gear, fishing location, number of fishers per boat, and the total numbers of boats out fishing on a given day (absolute effort).

[1]www.kobotoolbox.org

Monitoring was carried out daily where possible, at the time of day when most fishing boats returned to shore. Data collectors recorded the start time and end time of the trip, the gear used, the number of fishers on board, and the boat type. The habitat where fishing took place (reef, FAD, deep, mangrove, and beach) was recorded and was used to categorize trips to compare catch rates. Given the focus on comparing reef and FAD catch rates, in some comparisons deep, mangrove and beach catches were grouped as "other." The catch was quantified by the calculated weight and number of fish species present, the intended purpose of the catch (for food, sale, or both), and the current market price of each species (fisher's estimate).

Effort and Boat Activity Estimation

In this study, we use raw catch per unit effort (CPUE) to compare catch rates of different habitats across the different sites. Raw CPUE is simply the total catch divided by the sum of an observable measure of effort associated with the catch (Maunder et al., 2006). We standardize effort into the unit of fisher-hours

on each fishing trip, calculated as the trip duration in hours multiplied by the number of fishers onboard. The complexity of more robust CPUE standardization of multiple gears in tropical, multiple habitat, mixed species fisheries make it impractical in a livelihoods context.

In February 2018, solar-powered vessel tracking systems (VTS) developed by Pelagic Data Systems Inc., were installed on 50 boats across the four sites (Adara, N = 5; Vemasse, N = 15; Adarai, N = 15; Biacou, N = 15) to gather continuous boat location information at a frequency of 1 position every 10 s. The high-resolution tracks from these VTS from February to December 2018 enabled accurate quantification of fishing trip frequency and duration from which site-specific vessel activity coefficients (VAC) were derived, under the assumption that total monthly fishing effort does not change significantly within the year.

Total Production Estimation by Site

To estimate the total monthly catch for vessel-based SSF in each of the four sites, we combined observed landings data with VAC for paddle canoes and motor boats derived from VTS, and vessel census data from the Timor-Leste's MAF. The monthly catch (in tonnes; C) per site was calculated as:

$$C = \sum_{b} CPUE_b \times VAC_b \times N_b \times 0.001$$

where b refers to the boat type (canoe or motor-powered), *CPUE* is the monthly catch per unit effort in kg per hour, *VAC* is the average total fishing hours per month per vessel, N is the total number of active boats, and 0.001 converts the value from kg to tonnes. It is difficult to estimate the contribution of shore-based fishing to the total national catch since no census data are currently available for this fishing mode. However, shore-based fishing trips represented a minor proportion of trips recorded (∼4%), and the CPUE was substantially lower than the boat-based trips on average (data not shown), therefore we elected to restrict our analysis to boat- and canoe-based fishing only.

Effects of FADs on Catch Rates

To estimate the effects of FAD presence on catch rates, effort, and trip success during the deployment period generalized linear mixed models were fitted to explain the observed fishery production results. Three response variables were analyzed: CPUE in kg per fisher-hour (rounded to the nearest integer and modeled as a Poisson random variable), CPUE in number of fish per fisher-hour (also modeled using a Poisson distribution), and trip success rate (the proportion of trips with non-zero catch; modeled as a binomial response). CPUE values were modeled as Poisson-distributed variables in order to account for left-censoring in the data (on average 13% of the trips were recorded with no catch). For each analysis, the habitat type was included as a fixed effect (three levels; FAD, reef, or other) and site as a random factor (four levels). Temporal effects were not tested due to a lack of seasonal replication over time. Gear types were pooled due to insufficient replication and balance to account for the preferential use of certain gear types at certain habitats (for example, the preferential use of spear guns on reefs and seine nets only deployed on FADs). The final model equation is thus defined as:

$$y \sim X\beta + Zu + \varepsilon$$

where y is the vector of production values (kg per hour, no. of fish per hour, or success rate), X is the matrix of predictor variables, β is the fixed-effect regression coefficients (habitat type), Z is the design matrix for the random effects (site), u is the vector of random effects, and ε is the vector of residuals. Plots of residuals versus fitted values and normal quantiles were inspected for valid error distributions and variance homoscedasticity. Production analyses were carried out in the R environment (R Core Team, 2018) using the packages lme4 (Bates et al., 2015) and effects (Fox, 2003).

Cost-Benefit Analysis of Nearshore FADs

A site specific rate of 100% return on investment in days (*RoI*) is calculated based on the catch dividend of the FAD, not the total catch of the FAD. The multiplication of *Price* of fish ($/kg) by the change in catch rates between FAD and non-FAD fishing [$\Delta CPUE$ in kg/(fisher × hour)], returns the change in revenue per unit effort. S is the proportion of catch sold; VAC_{day} (average total hours fishing per boat per day) and N (total number of boats), are site specific values of effort; and F is the site specific frequency of FAD trips as a proportion of total trips (adapted from Sharp, 2012). The relatively short longevity of FADs negated the need for the inclusion of an annual discount rate to account for inflation.

$$RoI = \frac{Investment}{\left(\left(\left(VAC_{day} \times N\right) \times F\right) \times \left(\left(\Delta CPUE \times Price\right) \times S\right)\right)}$$

By basing the calculation of *RoI* only on the proportion of fish sold, our estimates are inherently conservative, given we are placing no value on fish caught for consumption. In reality, any increase in production from FADs would also provide significant food and nutrition security value at a household level. The *Price* of fish ($/kg) as sold by fishers was collected throughout the landings survey sampling period and the mean sale price from these data was used to estimate the difference in hypothetical revenue between FAD and non-FAD fishing. There were no discernible differences between mean prices per species across time, so a cumulative mean was used[2].

Catch Assemblages and Species Diversity

Species were identified by trained data collectors using a list of 130 known local species with photos. Unknown species were listed as such and submitted along with a photograph taken of the species. A further 36 species were added to the species list in this

[2]The demand for fish is still low in Timor-Leste, even in urban centers. Large pelagic fish do not sell well for reasons of lack of capital, lack of refrigeration, no export market and only limited local cold supply chains. The most popular species are those that are small and dry well (sardines, scads, and mackerels), but all species have very similar market values with small price fluctuations based on abundance than on species (large catches sell for cheaper so that fishers can offload it all) (López-Angarita et al., 2019).

way, using common and local names, or to class and order level, such as catfish, sharks, and stingrays. However, the full diversity of species landed is likely to be grossly underestimated given the lack of taxonomic research and knowledge in Timor-Leste.

As a measure of relative abundance or commonness of each species (i) in the catch composition, an index of relative importance (IRI) (Kolding, 1989) is used:

$$\%IRI_i = \frac{(\%W_i + \%N_i) - \%F_i}{\sum_{j=1}^{S}(\%W_j + \%N_j) - \%F_j} - 100$$

where $\%W_i$ and $\%N_i$ is percentage weight and number of each species (i) of total catch, $\%F_i$ is percentage frequency of occurrence of each species in the total number of samples, summed over all species from $j = 1$ to S.

Diversity and relative evenness of catch assemblages were compared by habitat and landing site using Shannon's Diversity (H) and evenness (J) indices, calculated in the R environment (R Core Team, 2018) using the vegan package (Oksanen et al., 2018). The %IRI was calculated using PasGear fisheries analysis software (ver. 2.11, Kolding and Skaalevik) (Kolding, 1999).

RESULTS

The gear types in use across sites were similar, with gill nets and hand lines predominating, however, in Biacou and Vemasse the use of specialized scoop seine net gear for FAD fishing was documented. The FADs deployed as part of this study were open access in all sites except Vemasse, where FADs were incorporated into existing FAD fishing groups that manage and fish at specific FADs, invest in gear, and hold shares of the catch. Catch success on FADs in Vemasse was reportedly sporadic, often catching nothing, and highly seasonal, but on good days the daily catch value could exceed USD $1000 (almost paying for the FAD costs in one day).

A total of 26.7 t of fish and invertebrates were recorded from 184 fishers on 3,277 fishing trips across the four sites between April 2017 and March 2018. Landings were comprised of at least 66 species. Catch rates were variable across space and time, with the highest overall mean catch rate of 2.78 kg/(fisher × hour) recorded from the community of Adara (**Table 2**). In terms of fishing effort, FAD fishing was only conducted year-round in Adara, whereas, at other sites, FAD fishing was restricted to certain seasons (**Figure 2**). In Biacou there were not sufficient data collected to assess fishing behavior year-round.

Effort and Boat Activity Estimation

The mean distance of deployed FADs from shore was 2 km. The average range (±SD) of motorized boats across all sites was 5.4 km ±2.9, and non-motorized boats was 1.6 km ±1.5. Across all sites, paddle canoes averaged 8.2 trips per month, and the average trip length was 3 h and 22 min, with a median effort of 4 fisher-hours (including fishing and traveling time). Motorized boats went out for an average of 4 h and 4 min per trip, with a median effort of 10 fisher-hours per trip and an average of 15.3 trips per month. Reef-fishing was associated with the shortest mean trips, with a mean duration of 2.93 (2.38/3.71) h per trip (Poisson mean with asymmetric 95% confidence bounds in parentheses). FAD fishing was similar in duration to other habitat types, accounting for 3.5 (2.83/4.42) h per trip and mangrove and deep sea for 3.4 (2.76/4.27) h per trip on average. Site-specific VACs are shown in **Table 2**.

Effects of FADs on Production

After accounting for random site variation, we observed a significant positive effect of FADs on productivity, with a mean CPUE value of 2.17 (1.84/2.54) kg/(fisher × hour) for FAD-associated fishing (Poisson mean with asymmetric 95% confidence bounds shown in parentheses), compared with 1.21 (0.97/1.51) kg/(fisher × hour) for reef fishing and 0.8 (0.68/0.93) kg/(fisher × hour) for other habitats (beach, mangrove, and deep sea; **Table 3**). Disaggregation by site revealed that this pattern was primarily driven by Adara and Vemasse, whose FAD-associated CPUE values were 2.8 and 5.3 kg/(fisher × hour), respectively (**Figure 3** and **Table 3**). A similar pattern was observed when considering CPUE in terms of the number of fish caught per fisher-hour. FAD-associated catches averaged 13.7 (9.8/19.1) fish/hour, which was higher than that observed for reef-fishing [2.49 (1.75/3.53)], or beach, deep sea and mangrove-fishing [9.2 (6.59/12.85)]. This was also primarily driven by Adara and Vemasse, with average CPUE values of 8 and 40.2 fish/hour, respectively.

The trip success rate was relatively constant across habitats and sites, with around 90% of trips producing a non-zero catch on aggregate for most of the sites. The exception was Biacou, where the trip success rate ranged from 54–76% depending on the habitat type. Here, reef fishing was generally the most likely method to experience a non-zero catch (76%), while FAD, deep, beach, and mangrove showed a similarly low trip success rates of 54–62%.

TABLE 2 | General catch statistics by landing site of four small-scale fishing communities in Timor-Leste.

Site	Diversity (Shannon's H)	Evenness (Shannon's J)	VAC [hr/(month × boat)]	Recorded catch (*N* fishes)	Recorded catch (kg)	Recorded effort (h)	Mean trip time (h)	CPUE [kg/(fisher × hour)]	Motorized vessels	Canoes	Total catch (t)
Adara	1.28	0.37	19.13	19288	6675	2729	2.5	2.78	3	12	28.4
Adarai	0.4	0.13	53.42	32875	3360	3210	3.21	0.97	12	60	44.3
Biacou	1	0.26	46.74	73302	5888	7367	4.32	0.92	41	8	72.5
Vemasse	1.16	0.43	76.24	106626	10761	5669	3.97	0.94	9	23	24.1

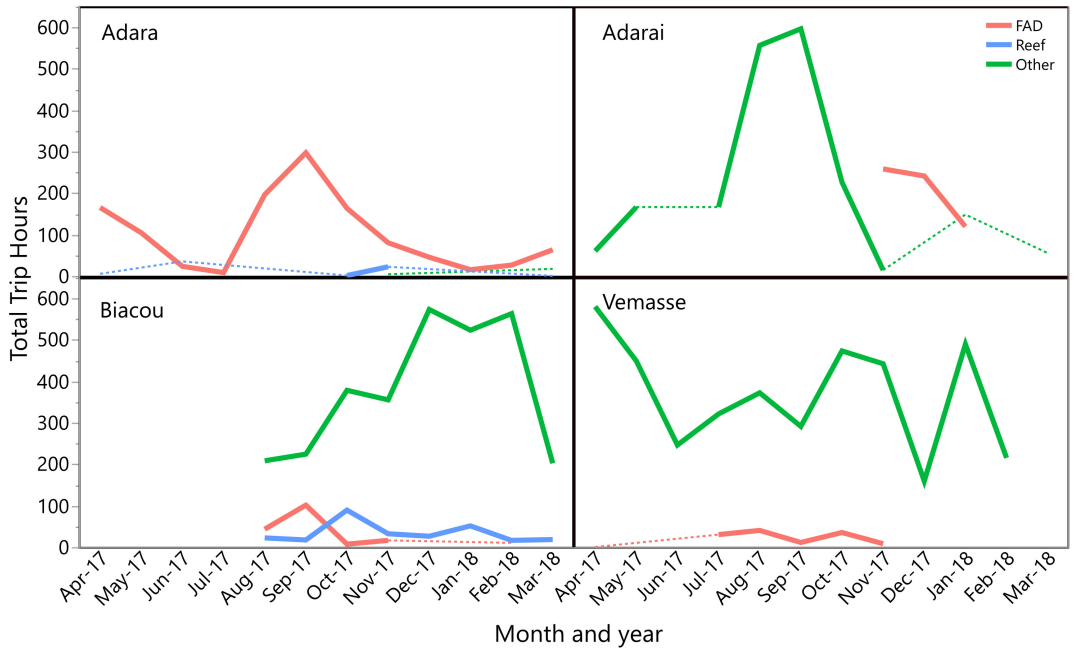


FIGURE 2 | The total number of fishing trips for which catch was recorded, by month and habitat at four sites in Timor-Leste. Solid lines represent continuous data. Connecting dotted lines reflect no data.

Total Production and Cost-Benefit Analysis of FADs

The highest estimated total annual production was seen in Biacou with 72.5 t, followed by Adarai with 44.3 t of fish landed during the 12-month study period (**Table 2**). The mean longevity of FADs deployed in this study was 11 months. The mean price per kilogram of fish landed was USD $2.49/kg, with slight variations by site (**Table 4**). Using an approximate FAD investment cost of $1250 to account for equipment, construction and deployment, and assuming consistent steady catch rates throughout the year, the time to 100% return on investment ranged from 18 days in Vemasse to 3343 days in Biacou (**Table 4**). However, based upon large variations between sites in terms of total effort, catch volumes, FAD fishing frequency and proportion of catch sold, the RoI varied significantly according to the site (**Table 4**). In Biacou, FAD fishing did not show significantly higher catch rates, so a RoI will likely never be realized in the lifetime of the FAD.

Catch Assemblage Species Diversity

Fish aggregating device catches across all locations was dominated by three species, the short-bodied scad (*Rastrelliger brachysoma*), the mackerel scad (*Decapterus macarellus*), and sardines (likely to represent a complex of up to five locally occurring species due to difficulties in identification), which comprised 96% (IRI) of landings. Adarai and Biacou are for the most part single species fisheries, with >90% of their catches comprised of *R. brachysoma* (99.6%) and sardines (*Sardinella* spp.) (92.1%), respectively. In Vemasse 92.2% of the catch was of two species (*D. macarellus*, 79.8% and *Sardinella* spp. 12.4%). Sardines are only found in catches from mainland sites. In Adara on Atauro Island, the top three species (*R. brachysoma, D. macarellus, and Pterocaesio tile* – striped fusilier) account for 98% of the catch (**Figure 4**).

Catch assemblages from reef fishing were much richer in species than from FAD fishing, with Shannon's diversity scores of 2.4 for reef compared to 0.98 for FADs (**Table 3** and **Figure 4**). The highest biodiversity in catch assemblages was seen from reef fishing in Biacou (2.48). Many of the species included in FAD catch assemblages can be accounted for by fishers' catches in reef habitat on the way back to shore from the FAD. Landings records were not of sufficient granularity to account for multiple habitats fished per trip.

DISCUSSION

This study shows that nearshore FADs can be a cost effective tool to increase catch rates of non-reef fish in rural coastal fisheries, without necessitating a transfer to new gears or methods. However, where more specialized fishing and gears were deployed, the results were even more pronounced. These results suggest that the incorporation of nearshore FAD deployments with fishery closures of nearshore reef areas, as employed in emergent marine resource co-management in Timor-Leste (Tilley et al., 2019), could be a potential way to mitigate common livelihood costs to fishers caused by displaced fishing effort (Cinner et al., 2014). However, to maximize the rate of return on investment, the selection of sites for FAD deployments should be carefully explored to account for social and ecological aspects. In the following sections, we explore the effect of FADs on fisheries production, biodiversity, and food security while evaluating the

TABLE 3 | Point estimates (Poisson means) of catch rates and 95% confidence bounds from FAD, reef, and other habitats (deep, mangrove, and beach) from four small-scale fishing communities in Timor-Leste.

Site	Habitat	CPUE [kg/(fisher × hour)]	CPUE (*N* individuals/hr)	Success rate	Recorded trips	Recorded catch (*N* individuals)	Recorded Catch (kg)	Diversity	Evenness
Adara	FAD	2.8(2.66/2.96)	7.99(7.74/8.24)	0.961(0.939/0.975)	485	18480	6413	1.24	0.37
Adara	Other	0.86(0.39/1.91)	3(1.96/4.6)	1(−−/−−)	7	163	41	1.05	0.96
Adara	Reef	1.25(0.92/1.7)	3.81(3.19/4.55)	0.875(0.711/0.952)	32	645	220	0.87	0.34
Adarai	FAD	1.16(1.02/1.32)	10.98(10.52/11.46)	0.959(0.919/0.979)	193	10794	1184	0.17	0.07
Adarai	Other	0.87(0.8/0.94)	9.15(8.92/9.38)	0.905(0.88/0.925)	682	22081	2175	0.47	0.16
Biacou	FAD	1.25(1/1.56)	6.92(6.29/7.61)	0.541(0.416/0.661)	61	2610	411	0.82	0.37
Biacou	Other	0.8(0.74/0.87)	12.48(12.22/12.75)	0.62(0.583/0.656)	669	69338	4675	0.75	0.21
Biacou	Reef	1.24(1.03/1.49)	2.28(1.99/2.61)	0.761(0.663/0.837)	92	1354	802	2.48	0.75
Vemasse	FAD	5.27(4.68/5.94)	40.18(38.47/41.95)	1(−−/−−)	51	59098	7900	0.08	0.11
Vemasse	Other	0.62(0.57/0.67)	12.29(12.07/12.5)	0.962(0.948/0.972)	1005	47528	2861	1.3	0.48
Total	FAD	2.17(1.84/2.54)	13.7(9.8/19.13)	0.926(0.808/0.974)	790	90982	15908	0.98	0.27
Total	Other	0.8(0.68/0.93)	9.2(6.59/12.85)	0.9(0.758/0.963)	2363	139110	9753	1.34	0.35
Total	Reef	1.21(0.97/1.51)	2.49(1.75/3.53)	0.939(0.83/0.98)	124	1999	1022	2.4	0.69

Poisson means with asymmetric 95% confidence bounds shown in parentheses.

opportunities and constraints of scaling FAD deployment in Timor-Leste with the aim of providing recommendations for a government-led, national level FAD programme.

Effects of FADs on Catch Rates and Effort

Results suggest that despite high site variability, fishing on FADs resulted in higher catch rates per unit of effort [kg/(fisher × hour)] than other habitat types. Greater amounts of fish being landed that can be distributed, is the first step toward improving rural access to fish and the critical micronutrients they provide. Timor-Leste waters are thought to be relatively unproductive, but average reef catch rates found here are similar to those reported in Fiji [0.9–1.6 kg/(fisher × hour)] (Kuster et al., 2005) and Apo Island in the Philippines [1–2 kg/(fisher × hour)] (Maypa et al., 2002). Additionally, nearshore FAD and non-FAD catch rates are comparable to neighboring countries in Asia-Pacific such as the Solomon Islands, where FAD fishing rates were reported from 1.04 to 2.96 kg/(fisher × hour) and non-FAD fishing from 0.87 to 2.16 kg/(fisher × hour) (Albert et al., 2014). Most studies on the effects of FADs on increasing catch rates are assessing FADs placed many miles offshore, targeting tuna, and thus are not applicable for an artisanal fleet consisting of mostly paddle canoes fishing on fringing reefs. There is far less information available on the effects of nearshore or coastal FADs [defined as within 12 nautical miles of shore (Gillett, 2016), but typically not more than 3 miles from the coast]. In Niue, nearshore FADs increased catch rates by only 27% over a 2 year period, compared to an offshore catch rate increase of 113% (Sharp, 2011). Other reported offshore annual gains reach even higher magnitudes, such as 340% in La Reunion (Detolle et al., 1998) and 1120% in Mauritius (Beverly et al., 2012).

Our findings support the positive FAD effects seen in other studies, but also confirm the significant variability between sites as seen in the Solomon Islands (Albert et al., 2014). Unfortunately due to relatively small number of data points and potential confounding of various factors affecting catch and effort (fishers utilizing habitat specific gear types, e.g., seine nets on FADs) we were unable to completely tease apart the effects of aggregation devices from variation among additional measured and non-measured factors, such as fishing gear type, substrate type, bathymetrical features, proximity to markets and human population centers, etc. Numerous factors affect catch rates, and there is no accepted justification for standardizing CPUE by the amount of fishing time when combining an active method, such as spearfishing, with a passive method, such as gill nets (Gibson-Reinemer et al., 2017). In using raw CPUE standardized by time, our study incorporates a significant statistical assumption in considering that the various gear types have equal catchability coefficients [the portion of the stock captured by one unit of effort (Maunder et al., 2006)], or even the same gear type is equal among fishers (e.g., using the same length gill net). However, the diversity and complexity of small-scale, mixed species fisheries make the adequate standardization of effort (see Maunder and Punt, 2004) for this type of study impractical. Furthermore, in a livelihood context, relevant considerations of "effort" are

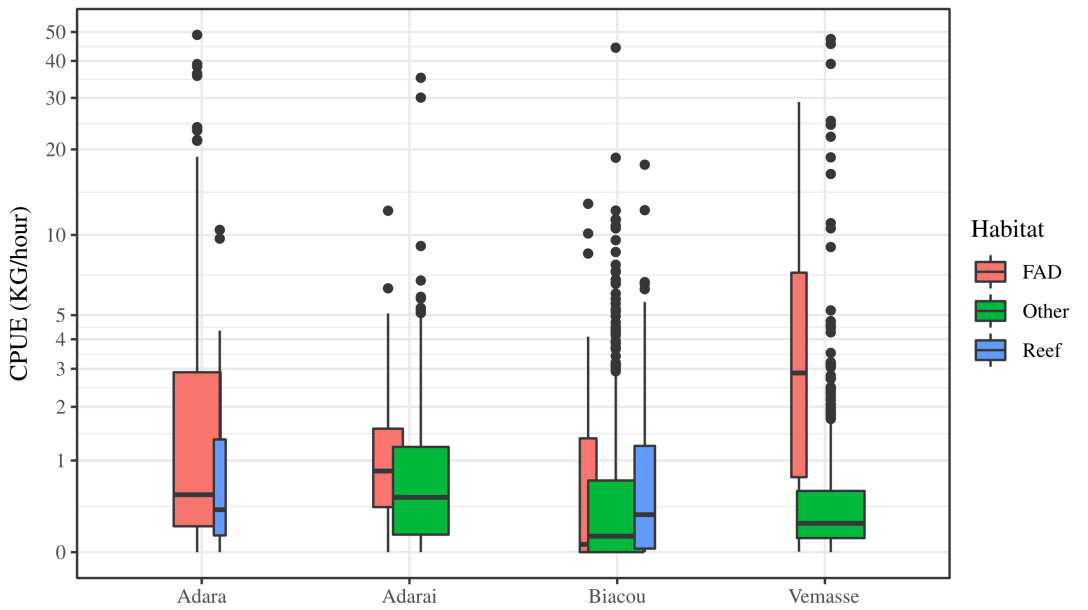


FIGURE 3 | Comparison of catch rates of FAD fishing, reef fishing, and other fishing across four sites in Timor-Leste using the weight of fish per fisher hour on a log axis (data untransformed). Trip records other than to FADs and reefs in Adara are not shown due to the low number of observations (n = 7). Box width is proportional to the number of observations for each site/habitat combination. Box midlines reflect the median values, box boundaries show lower and upper hinges (first and third quartiles – or 25th and 75th percentiles), and whisker limits are 1.5 × interquartile range. Outliers beyond these limits are shown as points.

energetic expenditure and time. We did not have the capability to measure energetic expenditure of fishers, so the key insight into comparable effort is the time expended (that could otherwise have been committed to alternative livelihood activities). Under these assumptions, our results indicate a positive net effect of FADs on overall catch rates, with corresponding positive effects on rates of return on FAD investment at local and regional scales.

The highest catch rates for FADs compared to other habitats in Timor-Leste were seen from the site of Vemasse. This community has been deploying traditional FADs of their own for the past few years and have created FAD fishing groups of ∼12 people who hold varying numbers of catch shares based on their investment in the equipment or labor. These FAD groups also utilize a specialized fishing technique documented in Mills et al. (2013) where lights are deployed above the FAD before dawn, then a modified scoop-seine net is deployed around the schooling fish. Even incorporating the additional investment costs of specialized seine net fishing gear (∼USD $1,000, Vemasse fisher pers. comm.), the time to 100% RoI would be ∼33 days. At the other three sites, fishers predominantly using gill nets and hand lines on FADs, just as in reef and open water areas, with some infrequent use of scoop seine nets in Biacou, but without the organized FAD group structure.

In Biacou, fishers traditionally focus on pelagic species. Analysis of vessel movements from GPS tracking show this pattern clearly, with the mean trip range (± SD) of vessels in Biacou reaching 7.5 km ± 3.4), which is more than double that of all other sites (Adara: 1.4 km ± 2.6; Adarai: 2.2 km ±1.3; and Vemasse: 3.5 km ±3.5) (**Figure 5**). This existent capacity to fish further offshore may account for no apparent differences between FAD fishing catch rates and other fishing, but the lack of year round data from Biacou make this inconclusive. The highly variable success of FADs seen in our results, suggests that the location of nearshore FAD deployments in Timor-Leste needs to be selected carefully based on gear types already in use and catch rates, and should incorporate the collection of some baseline catch monitoring from potential sites. These findings corroborate those of Albert et al. (2014) in the Solomon Islands, where villages presenting low catch rates, limited diversity of fishes, or degraded reef fisheries, were likely to benefit the most from access to a

TABLE 4 | Site-based Returns on Investment (*RoI*) in days for nearshore FADs at four sites in Timor-Leste.

Site	FAD trips/total trips (*F*)	Total Boats (*N*)	VAC_{day} (h)	Δ*CPUE*	*Price* of fish ($/kg)	Proportion of catch sold (*S*)	*RoI* (in days)
Adara	0.82	15	0.64	1.55	$1.81	0.37	154
Adarai	0.27	72	1.78	0.29	$2.19	0.69	82
Biacou	0.21	49	1.56	0.01	$2.80	0.83	3355
Vemasse	0.52	9*	2.54	4.65	$2.12	0.59	18

*Variables for the calculation of RoI are in bold, following Eq. 3. *In Vemasse only motorboats fished on FADs so canoes were excluded from the calculation of RoI.*

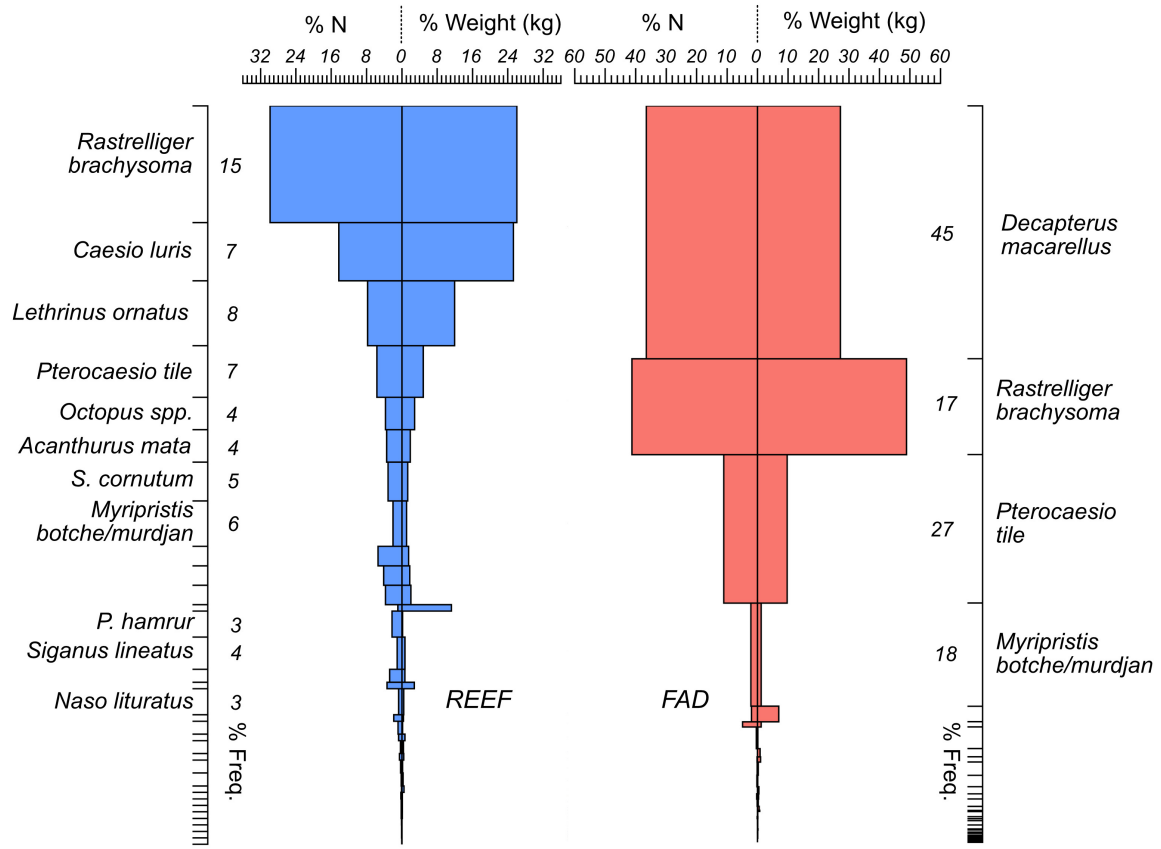


FIGURE 4 | Catch composition for small-scale fisheries landings from reef and FAD fishing from Adara, Atauro Island, Timor-Leste. The size of the box represents the %IRI of each species in each location, where %*N* is the proportion of total individuals, %Freq is the proportion of total samples, and %Weight is the proportion of total weight landed.

nearshore FAD. However, appropriate management steps should also be taken to ensure that this trend degradation does not merely transfer to pelagic fisheries, as improved catch rates can imply faster depletion of resources (Cabral et al., 2014).

Catch Assemblages and Biodiversity

Previous evaluations of sustained FAD programs suggest that in sites where pelagic resources are underutilized, they can bring considerable increases in fish yields through access to alternative resources (Beverly et al., 2012). The nearshore FADs studied here aggregate small, highly mobile pelagic fishes such as mackerels and scads, which are highly productive and can sustain relatively high levels of fishing pressure (Dalzell and Lewis, 1989; Dalzell, 1993). By providing access to an additional or more abundant source of fish, artisanal fishing effort may be reallocated from often heavily exploited coastal reef habitats (Beverly et al., 2012; Bell et al., 2015a). For example, catch rate increases in Mauritius were seen to coincide with a substantial and associated decrease in fishing effort in nearby lagoons, analogous to a reduction in pressure on reef systems (Beverly et al., 2012). Catch sampling was not sufficiently uniform to highlight changes in fishing effort on reefs, as the same fishers were not recorded every day. Fishing effort data before and after FAD deployments showed no significant increase or decrease on reef habitats specifically. However, redistribution of fishing effort in proportion to catch rates is a common phenomenon in SSF (Gillis et al., 1993; Gillis, 2003; López-Angarita et al., 2018; Peter and van Zwieten, 2018), suggesting that as fishers notice increasing catch rates at FADs, we are likely to see a geographical shift in effort.

A fisher focus group in Adara commented that the primary value of FADs for them was not larger catches, because there was a limit to what they could sell, but rather the reduced time taken to catch a sufficient quantity of fish, thereby allowing additional time to be dedicated to other livelihoods such as cultivating land and tending livestock. In a recent study from Timor-Leste, Mills et al. (2017) show that those households that fished year round had significantly better food security, and a lower number of livelihoods than seasonal fishers or farming households. FADs may have a very important role to play in this space, especially in areas of acute seasonal food shortages, to reduce vulnerability to shocks. Data collection on SSF and FADs in Timor-Leste is still ongoing. In future, we hope that a longer time series allows the testing of seasonality on fishing effort and catch rates.

Deploying nearshore FADs outside of designated no-take marine areas is a strategy that has been shown to contribute to marine protected area success in terms of recovery of



FIGURE 5 | Comparison of fishing behavior and trip range using tracks gathered by solar-powered vessel tracking systems by Pelagic Data Systems in four community sites around Timor-Leste between February and October 2018. The color gradient reflects the trip range, with the longest trips shaded in red and the shortest in white. The number of tracks represented in each pane are Vemasse 437, Biacou 1387, Adara 175, and Adarai 1256 (©Pelagic Data Systems. Satellite imagery©2018 HERE).

tropical marine habitat and recovery of fish stocks, as well as compensating for loss of fishing grounds for income of food for fishers (Gell and Roberts, 2003). In Adara, the community's revival of traditional practices of resource management called *tara bandu*, included the establishment of a closed area to fishing, where divers and snorkelers are charged a small entry fee (Mills and Tilley, 2017). According to community members, the accrual of this money for community development projects, and the regularity of fishing yields provided by the FADs have had a substantial effect on community livelihoods and wellbeing (Mills and Tilley, 2017). The emergence of co-management mechanisms of governance in Timor-Leste could leverage the potential of FADs even further to achieve combined conservation and sustainable development outcomes (Tilley et al., 2019).

Opportunities and Constraints to Scaling FAD Deployment

All the FADs assessed in this study were theoretically open access, to at least community level, and were all within 4 km of the shore to allow for all fishers to access – not just those with motorboats. However, a focus group discussion with fishers in Vemasse revealed the development of informal user-rights, where each FAD is "owned" by a group that has the exclusive rights to fish on it. This may account for the significantly higher catch rates and faster RoI in Vemasse (18 days) and highlights the potential importance of clearly defined boundaries in managing common resources (Ostrom, 1990), even if these are not necessarily legally recognized boundaries (Govan, 2009). Furthermore, this adheres to current ideas around best practice specific to FAD management (Beverly et al., 2012), where sustainable exploitation of resources is achieved through local stewardship and a sense of ownership.

By diversifying and enhancing the supply of fish, it is thought that fisher households will be more resilient to natural disasters, social and political instability, and climate variability (Bell et al., 2015a). This is because pelagic and reef species will likely respond very differently to broad-scale environmental changes or local pressures such as coastal development or reef degradation. Other suggested social benefits of FAD programs include increased employment opportunities, development of supporting industries, and improved nutrition (Sims, 1988). However, as FADs enable access to fishing areas further from the coast than traditional fishing areas, safety at sea becomes an important risk factor for fishers, as they

face different environmental conditions. Research regarding safety at sea in Timor-Leste is limited, but it has been reported that the major cause of accidents at sea is saltwater crocodile attacks followed by boat accidents due to sudden bad weather (Alonso et al., 2012). A survey conducted in 2012 showed that despite 97% of fishers identifying bad weather and big waves as a predominant problems faced at sea, 64% did not have safety equipment on board, mainly because it was overly expensive and considered unnecessary (Tsujimura et al., 2012). Given that there is currently no specific regulation on safety at sea for small-scale fishers in Timor-Leste, developing a legal framework alongside a FAD program will be important.

Nearshore FADs are not new to Timor-Leste. They are thought to have been introduced during the Indonesian occupation, and as such are already deployed independently by some communities on mainland Timor. On Atauro Island, FADs are notoriously difficult to deploy and maintain (Mills et al., 2013), due to strong currents and steep slopes to depths exceeding 4000 m. The average duration of FADs in the water across all locations was 11 months, which is significantly shorter than the lifespan of FADs in the Pacific, with the minimum expectation being 2 years, with some extending to 8 years (Sharp, 2011). However, this again is seemingly site and depth (**Table 1**) dependent, as the two FADs deployed on Timor-Leste′s south coast have been in place for 20 months as of December 2018.

In experimenting with FAD designs, the materials from which FADs were constructed by WorldFish and MAF were more expensive than might be utilized in a larger scale deployment program. Assuming equal efficacy at aggregating fish of low and high-cost FADs, this indicates RoI would be achieved even faster by reducing the initial investment cost. Beverly et al. (2012) suggest that quality should not be compromised because of funding and that a few well-made FADs are better than many of low quality that may be quickly lost. This, of course, makes economic sense in exploiting returns, and environmental sense in reducing the contribution of marine debris with broken and dislodged FADs (Macfadyen et al., 2009). However, there are instances when low-cost deployments may be preferable. If FADs are employed as a strategy by individual fishing groups in coastal communities (as opposed to a government program), they may lack the resources to purchase higher quality ropes and buoys. Furthermore, in Vemasse and Adarai, fishing on the FADs was only conducted seasonally when conditions were favorable, indicating a year-round FAD may be subject to wear and tear and be accumulating biofouling for a significant amount of time while it is not being fished. In this instance, low-cost FADs would be more appropriate.

Fish aggregating devices were not maintained in any organized or regular manner by fishers during this period. Maintenance programs, such as regular removal of barnacles and other biofouling organisms growing on the rope can reduce overloading of the buoys and increase FAD longevity (Beverly et al., 2012). At sites of increased FAD CPUE the average RoI was just 85 days, but intuitively, FAD longevity has a direct and significant effect on overall revenue generated. As with any financial investment, trade-offs of risks and gains scale with the number of shares, and in certain parts of Timor-Leste, the risks of losing FADs are significant. Hence, by encouraging and training fishers to act cooperatively in FAD fishing and management, individual risks to already vulnerable fishers with limited capital can be minimized, and benefits can be maximized.

CONCLUSION

- Near-shore deployed FADs for pelagic species can significantly improve artisanal catch rates and overall production at suitable localities in Timor-Leste; with specialized gear investment showing optimal return rates.
- Where a return on investment is seen, it is rapid and is indicative that government investment in FAD programmes would provide a substantial benefit to fishers and communities. FADs should not exceed 2 km from the coast to ensure paddle canoe fishers can access and thereby distribute benefits to the poorest fishers who are most likely to gain from even small improvements to catch rates.
- Given the heightened risk of accidents at sea related to fishing in FAD areas, the development of legal provisions that facilitate implementing measures for safety at sea is important. These should include increasing the capacity of the authorities, boat builders and fishers in safety measures, from boat building and maintenance to rescue operations.
- Fishers should be encouraged to form cooperative or group arrangements to reduce risk and improve livelihood resilience through better access to credit, product and insurance markets, and educational and training services.
- Catch rates improved by FADs could lead to faster depletion of resources in fish stocks of unknown biomass and productivity so communities supported by a FAD program should be encouraged and empowered to management through the collection of catch and effort data.

AUTHOR CONTRIBUTIONS

AT and DM designed the study. AT, MP, and SW collected the data. AT, SW, and JK analyzed the data. AT, SW, JK, DM, and JL-A wrote the manuscript.

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OPINION
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Bot Meets Whale: Best Practices for Mitigating Negative Interactions Between Marine Mammals and MicroROVs

Andrew Thaler[1,2]*, *E. C. M. Parsons*[3], *Asha de Vos*[4], *Naomi A. Rose*[5], *Courtney Smith*[6] *and Dominik Fretz*[7]

[1] Blackbeard Biologic: Science and Environmental Advisors, St. Michaels, MD, United States, [2] Center for Environmental Science, Horn Point Laboratory, University of Maryland, Cambridge, MD, United States, [3] Institute of Biodiversity, Animal Health and Comparative Medicine, Glasgow University, Glasgow, Scotland, [4] Oceanswell, Colombo, Sri Lanka, [5] Animal Welfare Institute, Washington, DC, United States, [6] Department of Environmental Science and Policy, George Mason University, Fairfax, VA, United States, [7] OpenROV, Berkeley, CA, United States

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Romuald Lipcius,
College of William & Mary,
United States

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Putu Liza Mustika,
James Cook University, Australia
Alice E. Hall,
Bournemouth University,
United Kingdom

***Correspondence:**
Andrew Thaler
andrew@blackbeardbiologic.com

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INTRODUCTION

Low-cost, portable, observation-class, underwater remotely operated vehicles (microROVs), which can be transported and operated by a single user, are increasingly common tools in scientific, industrial, commercial, and recreational ocean application. Over the last decade, the use of microROVs has boomed; four microROV manufacturers were poised to ship over 10,000 "underwater drones" in 2018 (Thaler, personal observation). This nascent industry provides an affordable underwater observation solution for marine science, conservation, education, and citizen science programs, as well as community groups and other stakeholders wishing to conduct independent marine environmental surveys and provides users with an opportunity to view marine wildlife with minimal disturbance (**Figure 1**).

This surge in the availability of microROVs also presents several new challenges to marine species. As more robots enter the water, often in the hands of inexperienced recreational users, there is increased potential for detrimental human/marine mammal interactions. MicroROVs are highly portable and have been identified as potential vectors for invasive species (Thaler et al., 2015). MicroROVs are also capable of causing harm to fragile marine ecosystems from contact with sensitive structures or tether entanglement. One possible outcome of increasing recreational use of microROVs is the increased harassment of marine mammals. The availability of new tools that allow people to approach and view marine mammals while maintaining their own safety has, if managed poorly, the potential to significantly alter the behavior of marine mammals (Higham et al., 2014; Smith et al., 2016). An example of this is provided by the whale and dolphin watching industry, which has developed rapidly world-wide, in some cases with demonstrably negative impacts on targeted populations (Bejder et al., 2006; Barragán-Barrera et al., 2017). Consequently, international policy bodies have been working toward a universal set of best practice guidelines for cetacean viewing over the past decade (e.g., Iñiguez, 2013; ACCOBAMS, 2016). Though not directly comparable, similar discussions have happened over the use of uncrewed aerial vehicles operated in close proximity to marine mammals (Thaler, 2014).

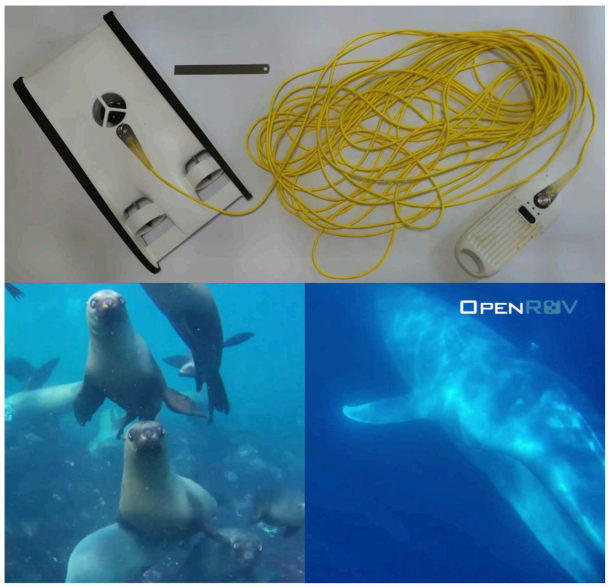


FIGURE 1 | Top: An example of a microROV system (OpenROV Trident) with 20 m tether and topside control system. 15 cm ruler provided for scale. Bottom left: Sea lions investigate a microROV off the California coast. Bottom right: a blue whale approaches and swims past a stationary microROV during a commercial whale watching excursion near Moss Landing, California. All photos used with permission.

To better understand the potential risks and to establish an anticipatory framework to minimize negative interactions between MicroROV operators and marine mammals, we, a group of six experts in microROVs and/or marine mammal tourism, conservation, and ecology, conducted a self-guided series of surveys to better identify the most likely and most damaging sources of harmful interactions between microROVs and marine mammals. We then established a set of best practice guidelines for the responsible operation of microROVs in the presence of marine mammals. Those guidelines, elaborated below, can be summarized as:

1. Educate users about the potential negative consequences of microROV operation in the presence of marine mammals.
2. Maintain situational awareness to avoid unintentional contact.
3. Maintain safe distances and avoid intentional contact.
4. Use microROVs as a tool to reduce the number of humans and large passenger vehicles on or in the water.
5. Avoid deployment where marine mammals are already active in an area.

METHODOLOGY

We deployed a highly-abridged version of the Delphi method, an iterative survey technique that aims to establish general consensus (Sumsion, 1998), amongst ourselves. A two-stage online survey, distributed among the six co-authors of this paper, was conducted using Google (Mountain View, California) software. This approach was implemented in order to identify broad agreement among co-authors, as well as critical points of disagreement and, as group discussion had to be coordinated across three continents, to establish an initial consensus framework that permitted more efficient discourse. While this approach does not produce a *de facto* "correct" answer, it does generate a reliable assessment of group opinion, from which consensus can emerge (Hasson et al., 2000).

In Survey 1, the authors were provided with an introduction to the iterative study design and asked to independently provide their assessment of the potential impacts of microROVs on marine mammals. We then ranked the likelihood of an impact occurring and the likelihood that those impacts would cause injury to, or behavioral change in, marine mammals using a five-point Likert scale. We also provided additional potential impacts.

In Survey 2, the authors reviewed the results of Survey 1 and considered whether we agreed with the emergent consensus. The outcome of these surveys was used as a launch point to further discuss and clarify the potential impacts of microROV operation in close proximity to marine mammals. Our conclusions and recommendations, however, represent a consensus expert opinion rather than empirically-tested observation and should be interpreted as such.

ETHICS STATEMENT

This research did not require Institutional Review Board approval. Participation was self-selected, and surveys did not include sensitive personal questions. As this process was implemented to assess consensus among the co-authors of this paper, anonymity could not be maintained.

POTENTIAL IMPACTS

For harassment that includes direct contact between marine mammals and microROVs, we considered collisions and tether entanglement most likely to occur, while ingestion was considered to be relatively less likely. If the microROV is lost due to tether breakage, it could be ingested if it is within an important feeding area, particularly for baleen whales that engage in feeding behaviors which have resulted in contact with unsuspecting humans at the surface (Pappas, 2019). Collisions were not expected to be as likely to cause injury as either entanglements or ingestion, and the size of the animal was expected to be a major determinant in the potential for injury as the relatively small mass of the microROV is unlikely to cause direct harm to an animal several hundred orders of magnitude larger than it.

For harassment that results in behavioral change through indirect contact with marine mammals, light and noise produced by the microROV were considered most likely to cause impacts and alter behavior. The mere presence of the microROV was also considered likely to alter behavior for some marine mammals, such as sea lions (*Zalophus* spp.), that are curious and may follow or investigate them (de Vere et al., 2017). Though there are no current studies on how light and noise produced by microROVs can impact marine life, one earlier

study demonstrated that the presence of a large research ROV had a detrimental effect on the feeding behavior of some animals (Spanier et al., 1994).

Although the impacts of operator behavior can result in both direct and indirect contact and harassment, we considered operator behavior as a separate category. Actions taken by the operator, such as whether or not to approach a marine mammal, are intentional impacts, rather than innate features of the equipment. Active, intentional harassment (e.g., pursuit/chasing of marine mammals) facilitated by access to microROVs was considered likely to occur and likely to result in both injuries and behavioral changes. Increased density of both boats and equipment in the water was also considered to have a high potential for impact. The presence of the microROVs was also considered likely to increase habituation to people, resulting in negative behavioral changes. Operator actions can also result in unintentional impacts, such as a marine mammal becoming entangled in the tether.

We collectively agreed that as microROVs become more available, they are more likely to be in the hands of untrained users, which can confound best practices and requires general user education. In addition, we determined that, while it may seem as if a certain number of boat lengths [a standard measuring tool used by marine mammal researchers (Dawson et al., 2008)] is the appropriate distance to remain from a marine mammal during directed activities, the submerged microROV may be much closer. This scenario could present challenges for enforcement officers, who can observe human behavior on the surface but may not be able to track submerged equipment.

GUIDELINES AND DISCUSSION

Based on the identified risks, we established a set of guidelines for microROV operators to minimize their potential impact on marine mammals. While several of these recommendations mirror existing wildlife viewing regulations that protect marine mammals in jurisdictions such as the exclusive economic zone of the United States, we have intentionally structured these guidelines to represent consensus best practices regardless of the regulatory environment in which the microROV user operates.

Education

Central to any mitigation strategy involving diverse stakeholders, ranging from professional to recreational, is user education. The following are critical to establishing a responsible user community: Ensuring all potential microROV users (1) not only understand the laws and regulations for wildlife viewing that apply to the jurisdiction in which they are operating, but understand why those regulations are in place; and, most importantly, (2) have internalized a stewardship ethic that motivates them to respect the rationale behind those regulations even when operating in regions where those regulations are not enforced. This is most effective when it occurs at point-of-sale or registration of the microROV. Thus, while the additional four guidelines relate to the user, this first one relates to the manufacturer. To most effectively convey the potential harm that microROVs could pose to marine mammals, the manufacturers are best positioned to educate their user base by providing informational material with each microROV sale.

Avoid Unintentional Contact by Maintaining Situational Awareness

As some of the most disruptive outcomes of interaction between marine mammals and microROVs are unintentional contact, users must maintain comprehensive situational awareness of their operating site, the location of their tether, and the presence of any marine mammals. When an animal-initiated approach is observed, users should first confirm that the microROV tether is not in the path of approach and then either remain stationary with thrusters powered down until the animal passes or remove the vehicle from the water while causing minimal disturbance.

Avoid Intentional Contact by Maintaining a Safe Distance and Piloting Responsibly

When operated in close proximity to marine mammals, microROVs should be treated no differently than any other vehicle. Intentional contact with marine mammals is not only highly disruptive but is illegal in some countries (Kindt and Wintheiser, 1985). MicroROV operators should familiarize themselves with local wildlife viewing regulations, always maintain a safe distance (50 to 100 m; distances can be estimated, where water visibility allows, by placing highly visible markers on the tether to act as a scale) when piloting a microROV in areas where marine mammals are present and maintain constant awareness over the location of both the robots and marine mammals. Where local regulations or professional standards exist for local tourism, microROVs should not get closer than the distances stipulated by local marine mammal approach standards. Any direct contact between a microROV and a marine mammal should be treated as an unacceptable encounter and microROV operations should cease immediately. Maintaining a safe distance will also mitigate the impact of noise produced by the microROV.

Treat microROVs as a Tool to Reduce, Rather Than Increase, Vehicle Density

MicroROVs present a powerful opportunity to allow a large number of people to safely view marine mammals. Because of this, it may be tempting to deploy multiple microROVs in regions where marine mammals are known to aggregate, thus increasing the risk of contact and behavioral alterations to the target species. MicroROVs should be treated as tools to reduce vehicle density by allowing multiple operators and viewers to use a single microROV feed as an alternative to many divers in the water or numerous tour boats. As multiple microROVs operating in a small area also create hazards for the devices, operators should adopt standards and protocols (such as "first-come, first-served" commonly used at popular SCUBA diving locations) to minimize microROV density.

Minimize Deployment in Regions of Known Ecological Importance to Marine Mammals

There are a number of locations that have been designated as Marine Protected Areas or identified as "hotspots" for marine mammals, year-round and seasonally, such as Kealakekua Bay in Hawaii (spinner dolphins; *Stenella longirostris*; Timmel et al., 2008; Tyne et al., 2015; Heenehan et al., 2017) or Samaná Bay in the Dominican Republic (humpback whales, *Megaptera novaeangliae*; Mattila et al., 1994; Betancourt et al., 2012; Gleason and Parsons, 2018). In addition, there are locations where marine mammals are reliably sighted and engaged in normal behaviors, such as foraging, mating, or nursing (e.g., California sea lions, *Zalophus californianus*, in the Channel Islands, and northern elephant seals, *Mirounga angustirostris*, at Año Nuevo State Park, both in California; Heath and Perrin, 2009; Hindell and Perrin, 2009). Operators should refrain from deploying microROVs in such areas, as well as in areas where marine mammals are present in large numbers. In cases where the microROV is already in the water, operators should recall the device if it becomes likely that direct or indirect contact could occur. Operators should always refrain from pursuing or otherwise interacting with marine mammals. In cases where rare and vulnerable species are observed (and particularly when engaged in critical behaviors such as forging, mating, or nursing young), microROV users should make every effort to remove their equipment from the water without causing additional disturbance.

CONCLUSION

Low-cost microROVs present tremendous opportunities for marine research, conservation, exploration, and recreation. With these opportunities comes a responsibility to ensure that microROV users minimize harm to the marine environment. As this nascent industry is still largely unregulated, it is up to the users and manufacturers to promote responsible operation of microROVs around marine mammals. By following this preliminary and evolving set of guidelines, microROV users can mitigate or minimize potential harmful interactions with marine mammals from the outset of microROV use expansion. We encourage all microROV users to incorporate these guidelines into their dive operations and encourage microROV manufactures to provide educational materials emphasizing these guidelines at point-of-sale for their customers.

AUTHOR CONTRIBUTIONS

AT and DF conceived the study. AT designed and implemented the survey. AT, EP, AdV, NR, CS, and DF participated in discussion, guideline development, manuscript drafting, and manuscript revisions.

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BRIEF RESEARCH REPORT
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Tracing Coral Reefs: A Citizen Science Approach in Mapping Coral Reefs to Enhance Marine Park Management Strategies

Chai Ming Lau[1], Abdul Adzis Kee-Alfian[2], Yang Amri Affendi[3], Julian Hyde[1], Alvin Chelliah[1,4], Yun Sing Leong[1], Yin Lee Low[1], Putri Asma Megat Yusop[5], Ven Thye Leong[1], A. Mohd Halimi[4], Yaman Mohd Shahir[4], R. Mohd Ramdhan[6], Ai Gaik Lim[7] and Noor Ikhwanie Zainal[7]*

[1] Reef Check Malaysia, Kuala Lumpur, Malaysia, [2] SEAlutions Sdn Bhd, Kuala Lumpur, Malaysia, [3] Institute of Ocean and Earth Sciences, University of Malaya, Kuala Lumpur, Malaysia, [4] Tioman Marine Conservation Group (TMCG), Reef Check Malaysia, Kuala Rompin, Malaysia, [5] Institute of Oceanography and Environment, Universiti Malaysia Terengganu, Kuala Terengganu, Malaysia, [6] B&J Diving Centre, Kuala Rompin, Malaysia, [7] Department of Fisheries, Ministry of Agriculture and Agro-based Industry, Putrajaya, Malaysia

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Edited by:
Edward Jeremy Hind-Ozan, Department for Environment, Food and Rural Affairs, United Kingdom

Reviewed by:
Jean-Luc Solandt, Independent Researcher, Ross-on-Wye, United Kingdom
Chi Chiu Cheang, The Education University of Hong Kong, Hong Kong

***Correspondence:**
Chai Ming Lau
edmund@reefcheck.org.my;
e_lcm@yahoo.co.uk

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Effective marine park management and protection of coral reefs can only happen if managers have adequate knowledge of reef health and area. However, obtaining such information is labor intensive and difficult with limited funding and time. Reef Check Malaysia was engaged by Department of Marine Parks Malaysia to map the coral reefs surrounding Tioman Island Marine Park and document health status and site specific threats. To achieve this, we utilized the Reef Check survey method, a simple, rapid and holistic standardized reef monitoring protocol based on scientific principles. This method is suitable where funds and time are limited. A total of 95 sites surrounding Tioman Island were surveyed with the assistance of certified Reef Check EcoDiver volunteers and representatives from local stakeholders. This citizen science approach proved successful and generated a baseline map revealing a difference in the health of coral reefs between the west and east sides of Tioman Island, where the West had <25% live coral cover as compared to >50% on the East. Combined with data on indicator fish and invertebrates, as well as human and natural impacts, the results suggest that Tioman Island should be separated into three distinctive conservation priority zones to enhance management strategies of this marine park. This is an example of an innovative way to engage and involve local stakeholders in planning conservation and management strategies.

Keywords: citizen science, marine protected area management, reef check, coral reef mapping, geographic information system

INTRODUCTION

Coral reefs around the world are facing growing threats from changes to the environment through climate change (Praveena et al., 2012; Rinkevich, 2015). Mass coral bleaching events have become more frequent, affecting reefs worldwide (Tun et al., 2010). Many scientists have called for more frequent monitoring of coral reefs in order to better manage this crisis (Tun et al., 2010). There is

also a need to improve mapping of coral reefs which will contribute to a better understanding of connectivity among networks of MPAs. Knowing the extent of reefs in an area can also help to focus management activities, especially in managing the limited resources available.

Around the world, many researchers and scientists have engaged non-scientist volunteers who are able to assist in data collection and compilation for scientific projects (Bonney et al., 2014). Such citizen science programs are very much underutilized (Cigliano et al., 2015) although information from such programs can be of great contribution to projects that have limited resources available. Reef Check is a citizen science tool created to include the participation of non-scientists in a scientific monitoring exercise (Hodgson, 1997, 1999, 2001; Hodgson and Stepath, 1998; Hodgson and Wilkinson, 2001). There are many monitoring and scientific survey exercises that use Reef Check data for management and better understanding the changes that are rapidly occurring in many reefs, especially in this region (Hodgson and Wilkinson, 2001; Hill and Wilkinson, 2004; Wilkinson, 2008; Wood and Dipper, 2008; Hughes et al., 2010; Wetzelhuetter et al., 2014). This paper elaborates on baseline data collected for coral reef status in Tioman utilizing the Reef Check method. Surveys were conducted with the assistance of local community stakeholders and volunteers, who represent the "citizen science" part of the program.

Tioman is one of the most popular tourist destinations in Malaysia, receiving some 250,000 visitors per year (Department of Marine Park Malaysia, 2017). With 81 resorts and 36 dive centers, mostly congregated on the western side, Tioman has become one of the top destinations for marine recreational activities. Tioman has been a marine protected area (MPA) with a no-take approach since being gazetted in 1994. After more than 20 years, there was a need to review the MPA approach and designating zones within MPA is one of the new approaches being considered. However, without a clear coral reef map of Tioman, it is difficult to assign zones. Therefore, the objectives of this work are (1) to develop a baseline coral reef map along the Pulau (= island) Tioman coastline, (2) to identify priority areas of coral reefs for effective management and (3) to demonstrate the importance of citizen science involvement in providing valuable management information.

METHOD

Study Sites

This study was conducted at Pulau (= island) Tioman, Malaysia. The island is situated between 02°48′52.1″ N, 104° 10′29.3″ E which is approximately 32 km off the east coast of Peninsular Malaysia, in the state of Pahang. The waters surrounding the island extending to 2 nautical miles from the coastal line were gazetted as a Marine Park in 1994 under the Fisheries Act, 1985.

For this study, a total of 95 sites were surveyed along the island's coastline of approximately 55 km with a distance of 500 m between each transect points. The surveys were conducted over 32 days by a team of surveyors comprising marine biologists and volunteers (both local islanders and others).

Reef Check Method With Minor Modification

This survey was conducted using the Reef Check protocol, which utilizes the concept of indicator species and is an eco-holistic approach in which three components of the reefs are recorded, namely fish, invertebrates and substrate (Hodgson, 1999). At each site, a 100 m transect is laid perpendicular to the shore. The transect is separated into four segments of 20 m with a 5 m gap in between segments. For both fish and invertebrates, assessments are conducted as a 5 m wide belt transect along each segment. For substrate, a Point Intercept Transect (PIT) is conducted at 0.5 m intervals along each of the segments with the assistance of a plumb line to remove observer bias. The benthic category touched by the plumb at each data point is recorded. The depth of the reef along the 100 m transect is recorded at the start and end points of every segment.

For the purpose of this survey, an additional category was added for a better representation of the study objectives. Dead coral (DC) is listed as an additional category instead of being categorized as rock (RC) under the standard 10 substrate categories in Reef Check.

Quality Control

All volunteers have been trained and certified following the standard training procedure of Reef Check Eco-Diver to ensure a minimum 80% correct identification of indicator species and substrate categories. All Eco-Divers were also required to have good diving skills, particularly buoyancy.

Data Analysis and Mapping

The data for substrate, fish and invertebrates were used to plot the baseline map, each group of indicators being classified into different categories. The Coral Reef Health Criteria (Chou et al., 1994) were used for the classification of substrate. The percentage of live coral cover (LCC), which is the sum of percentage of hard coral (HC) and soft coral (SC) were separated into four classes, each with a different rating, as follows:

(a) 0–25%: Poor;
(b) 26–50%: Fair;
(c) 51–75%: Good;
(d) 75–100%: Excellent.

The dead coral cover (DCC) was also subjected to the same class, however, with an inversed rating from LCC. The data of LCC and DCC formed the basis of the geographic information system (GIS) mapping using QGIS software for visual interpretation. As for fish and invertebrate indicators, only Parrotfish and Long-spined Sea Urchins (Diadema) were selected and plotted onto the map, respectively. The abundance of these two organisms were classified into classes based on other studies as a metric of healthy population (Carreiro-Silva and Mcclanahan, 2001; Bonaldo et al., 2017). For Parrotfish, abundance that is <5 ind.500 m^3 is considered unhealthy population while abundance with >15 ind.500 m^3 is considered healthy. For LSU, abundance with <6 ind.100 m^2 is healthy, abundance of 6 to 60 is moderate while abundance that is >60

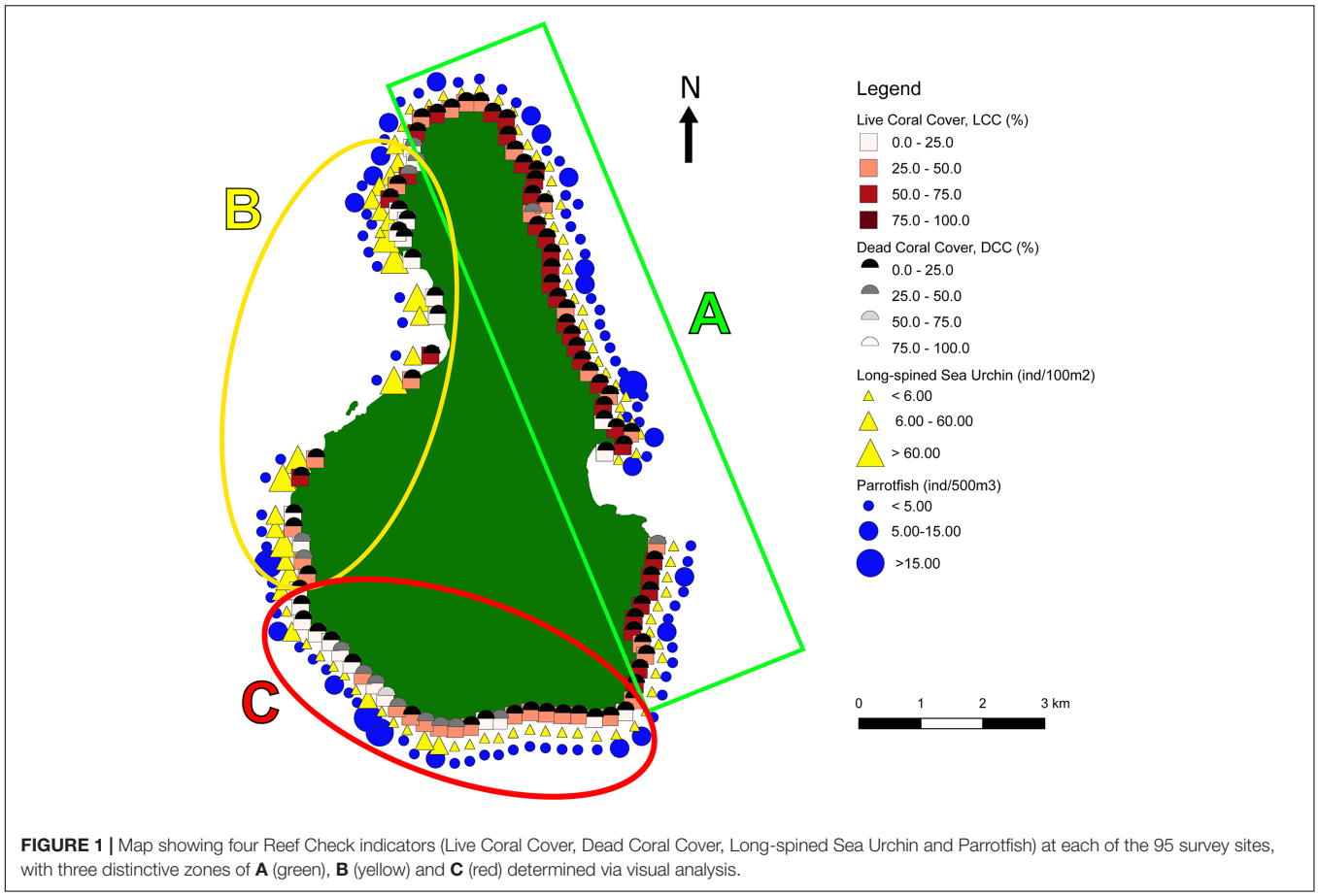


FIGURE 1 | Map showing four Reef Check indicators (Live Coral Cover, Dead Coral Cover, Long-spined Sea Urchin and Parrotfish) at each of the 95 survey sites, with three distinctive zones of **A** (green), **B** (yellow) and **C** (red) determined via visual analysis.

ind.100 m^2 is deemed unhealthy. All classifications and their respective values are presented on the map in **Figure 1**.

The percent cover of LCC and DCC as well as abundance of Parrotfish and Diadema were compared with a non-parametric Kruskal–Wallis test in which the fixed factor were the zones established from the visual interpretation of the GIS plotted map. When differences were significant, a *post hoc* Dunn test with Bonferroni adjustment method were used. All statistical analysis was conducted using R (R Core Team, 2019).

RESULTS

Substrates

Our survey estimated that Tioman is surrounded by an area of 5.46 km^2 of fringing reef. The biotic substrate composition of this area of coral reef is comprised of 32.3 ± 1.7% (mean ± SE) of HC, 6.4 ± 0.7% of SC, 2.2 ± 0.3% of nutrient indicator algae (NIA) followed by 1.0 ± 0.2% of sponge (SP) and 0.6 ± 0.1% of others (OT). Meanwhile, the abiotic substrate composition of the coral reef is 0.4 ± 0.1% of recently killed coral (RKC), 15.5 ± 1.0% of DC, 8.2 ± 0.8% of rock (RC), 7.0 ± 0.9% of rubble (RB), 20.8 ± 2.0% of sand (SD) and 5.7 ± 1.5% of silt (SI) (**Figure 2**). The average percentage of LCC for Tioman determined from this study is 38.8 ± 2.0% which falls into the rating of "Fair" under the Coral Reef Health Criteria.

Fish Indicators

There are nine indicator fishes listed under the Reef Check method and in this study, Butterflyfish (Chaetodontidae) topped the list in terms of abundance at a mean of 6.65 ± 0.53 ind.500 m^{-3} (mean ± SE). The second highest abundance is Parrotfish (Scaridae) observed at a mean of 4.33 ± 0.67 ind.500 m^{-3} followed by Snapper (Lutjanidae), Grouper (Serranidae) and Bumphead Parrotfish (*Bolbometopon muricatum*) at 0.92 ± 0.20, 0.71 ± 0.11 and 0.07 ± 0.04 ind.500 m^{-3}, respectively. The lowest abundance of fishes recorded are Moray eels (Muraenidae) and Sweetlips (Haemulidae) at only 0.05 ± 0.01 and 0.03 ± 0.01 ind.500 m^{-3} while Humphead Wrasse (*Cheilinus undulatus*) and Barramundi Cod (*Cromileptes altivelis*) were not found at all during the entire survey (**Figure 2**).

Invertebrate Indicators

Out of the nine Reef Check indicator invertebrates, only five were observed during this study. The highest abundance observed is the Long-spined Sea Urchins (LSU; comprised of the genus Diadema and Echinotrix) with an average of 15.02 ± 3.61 ind.100 m^{-2} (mean ± SE) followed by Sea Cucumbers (only three

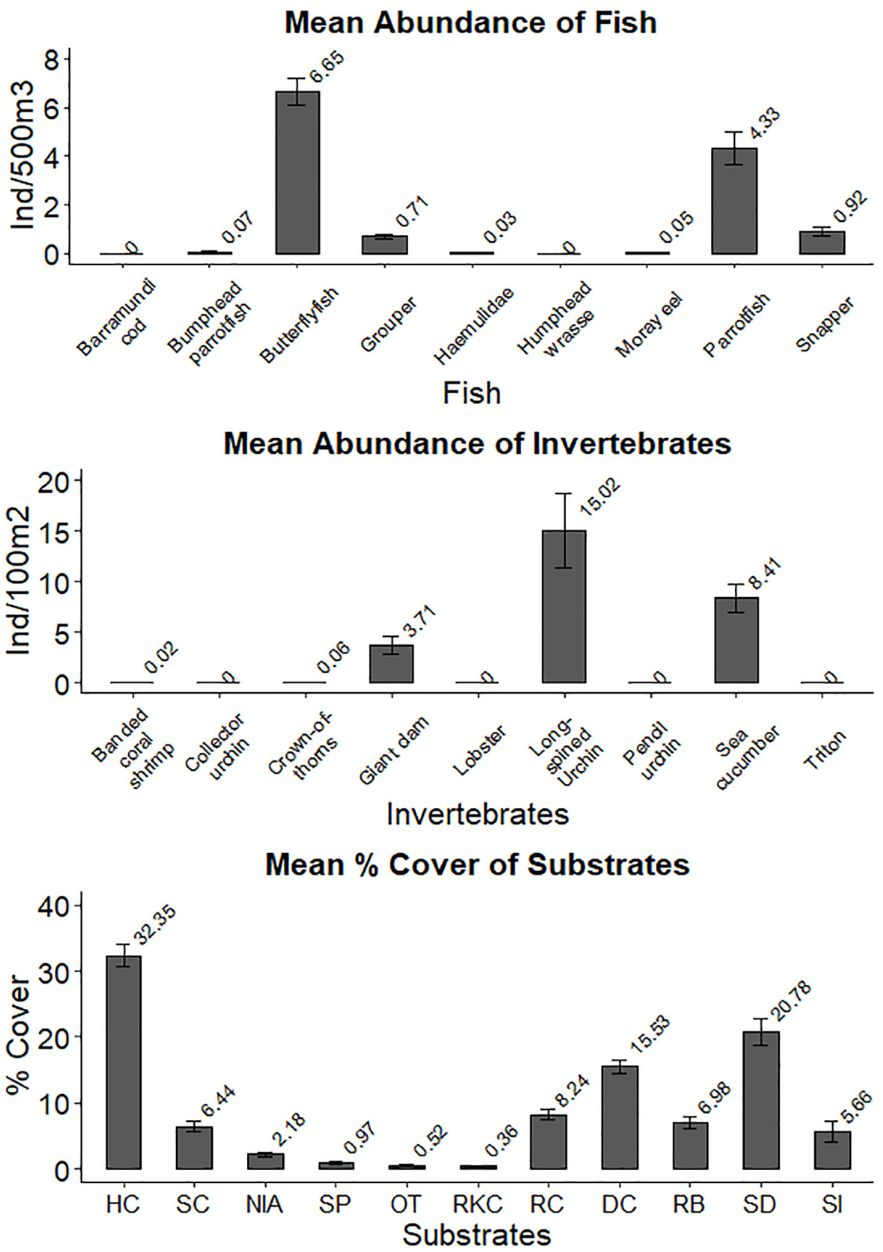


FIGURE 2 | Mean abundance (mean ± SE) of all Reef Check indicators for fish (individual/500 m^3) **(top)**, invertebrates (individual/100 m^2) **(middle)** and percentage coverage (mean ± SE) of Reef Check substrate categories (%) **(bottom)** for Pulau Tioman (n = 95).

species taken into account namely *Holothuria edulis*, *Stichopus chloronotus* and *Thelenota ananas*) and Giant Clams (*Tridacna* spp.) with 8.41 ± 1.39 and 3.71 ± 0.87 ind.100 m^{-2}. Only 0.02 ± 0.01 individuals of Banded Coral Shrimp (*Stenopus hispidus*) were observed within 100 m^2 while Crown-of-Thorns Starfish (*Acanthaster planci*) comprised 0.06 ± 0.02 ind.100 m^{-2} (**Figure 2**).

Map

The map shows the different classes for the four main indicators namely LCC, DCC, long-spined urchin (LSU) and Parrotfish at each of the 95 survey sites (**Figure 1**). The different color tones of square and semi-circle symbols indicate the different classes for LCC and DCC, respectively. For LSU and Parrotfish, the different sizes of triangle and circle symbols indicate the respective different abundance classes. For the purpose of this study, the overall reef health is represented by a combination of all four indicators. Sites with higher percentage of LCC and lower percentage of DCC as well as higher abundance of Parrotfish and lower abundance of LSU are considered to be healthy. In contrast, sites with low LCC, high DCC, low abundance of Parrotfish and high abundance of LSU are considered to be less healthy.

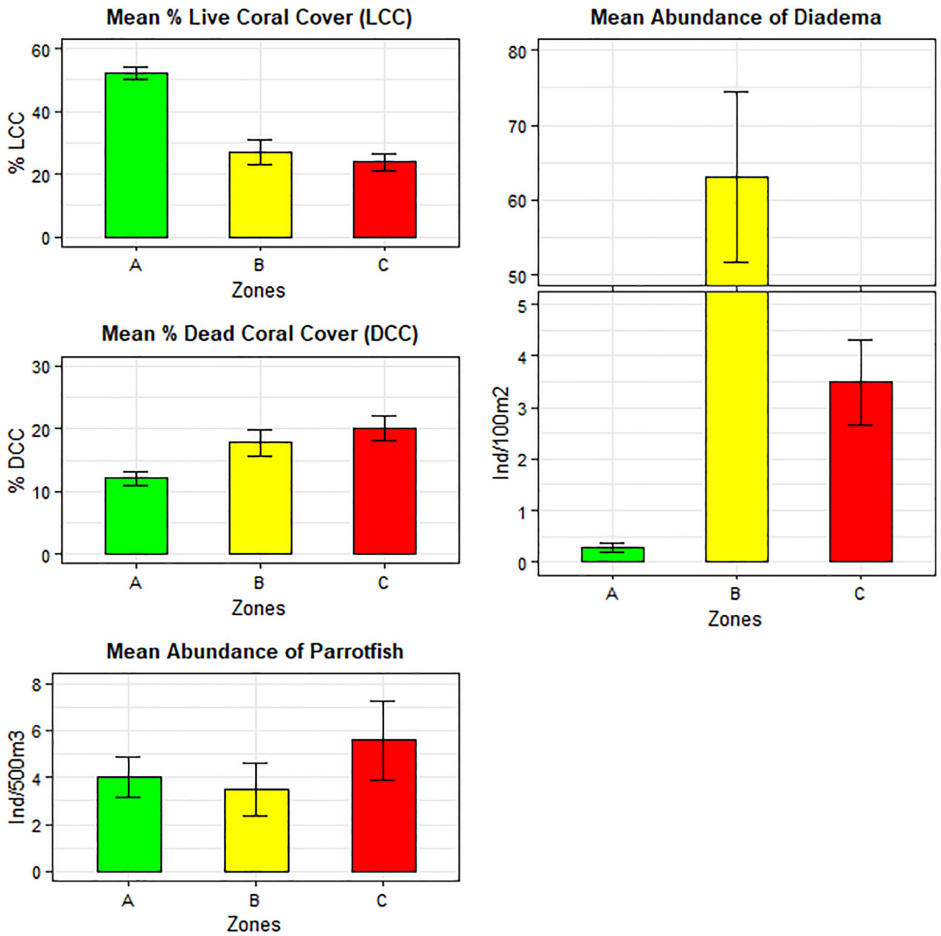


FIGURE 3 | The percentage coverage (mean ± SE) of Live and Dead Coral Cover (% cover), and abundance (mean ± SE) of Parrotfish (individual/500 m^3) and Diadema (Long-spined Sea Urchin) (individual/100 m^2) in the three respective zones (A, B, and C) determined in this study.

Zoning

The zoning on the map is determined using visual analysis on the classification of LCC and DCC, followed by LSU and Parrotfish. A clear pattern can be observed for LCC, DCC and LSU but not so for Parrotfish. Most of the sites with LCC in the 50–75% class or "Good" rating congregated at the north and east of Tioman. Meanwhile a majority of the sites with LCC rated as "Fair" and "Poor" (classified <50%) clearly congregated at the south and slightly scattered on the west. Sites with high DCC (>50%) are congregated on the south. LSU have a different pattern where sites with high abundance (>60.00 ind.100 m^{-2}) are congregated almost exclusively only on the west, whereas all other sites have low abundance (<6.00 ind.100 m^{-2}). There is no clear pattern observed for Parrotfish as sites with the highest abundance are scattered around Tioman. Hence, following the visual analysis of the patterns of LCC, DC, LSU on the map revealed that Tioman can be distinctively grouped into three zones namely A, B, and C (**Figure 1**).

The coral reef in zone A has a mean LCC of 52.2 ± 2.0% (mean ± SE) and low mean DC cover of 12.1 ± 1.0% (**Figure 3**). This zone also comprises sites with low abundance of LSU at 0.28 ± 0.08 ind.100 m^{-2} (mean ± SE). In zone B, the coral reefs have a mean LCC of 26.8 ± 3.8% and a few sites with moderate cover of DC (17.8 ± 2.2%). Zone B also comprised of sites with high abundance of LSU (63.00 ± 11.14 ind.100 m^{-2}). On the south of Tioman island, namely zone C, the mean LCC is in "Poor" condition (23.8 ± 2.5%) with several sites recorded high percentage of DC, 20.1 ± 2.0%. Additionally, the sites in this zone have an average low LSU abundance (3.49 ± 0.81 ind.100 m^{-2}). The abundances of Parrotfish are almost equal in all three zones, with Zone B having the lowest abundance of 3.48 ± 1.09 ind.500 m^{-3} (mean ± SE), followed by Zone A with 4.02 ± 0.84 ind.500 m^{-3} and Zone C with 5.58 ± 1.65 ind.500 m^{-3}.

Statistical analysis showed that all groups of indicator with the exception of Parrotfish have significant difference between zones ($p < 0.05$). *Post hoc* Dunn test reveal that for LCC, there was significant difference between zones A–B and A–C but not between zones B–C. This is followed by DCC, with significant difference only found between zones A–C. Meanwhile the difference of LSU between all zones were significant (**Table 1**).

TABLE 1 | Summary of *post hoc* Dunn test (with Bonferroni adjustment) of three survey components namely live coral cover (LCC), dead coral cover (DCC), parrotfish, long-spined urchins (LSU) comparing between zones A, B, and C.

Indicators	Zones	*Z* value	*P* unadjusted	*P* adjusted
LCC	A–B	4.870218	$1.114753e^{-06}$	**$3.344260e^{-06}$**
	A–C	5.923989	$3.142246e^{-09}$	**$9.426739e^{-09}$**
	B–C	0.573658	$5.661992e^{-01}$	$1.000000e^{+00}$
DCC	A–B	−2.3525603	0.0186446688	0.055934007
	A–C	−3.5511111	0.0003836085	**0.001150825**
	B–C	−0.8493834	0.3956679941	1.000000000
LSU	A–B	−7.855790	$3.972616e^{-15}$	**$1.191785e^{-14}$**
	A–C	−3.937606	$8.229843e^{-05}$	**$2.468953e^{-04}$**
	B–C	3.737330	$1.859846e^{-04}$	**$5.579538e^{-04}$**
Parrotfish	A–B	0.9607672	0.33666924	1.00000000
	A–C	−1.5375523	0.12415811	0.37247434
	B–C	−2.1328647	0.03293583	0.09880749

Zones with significant difference are shown in bold.

DISCUSSION

The citizen science method, Reef Check, was designed to be ecologically holistic with scientific principles (Hodgson, 1997) and has been applied in many research and conservation based projects (Hagan et al., 2007; Wilkinson, 2008; Hughes et al., 2010; Liu et al., 2012; Wetzelhuetter et al., 2014). Hence, the map produced provides reliable baseline information adequate for management to prioritize important zones for protection. Our project findings suggest that the management authority can utilize this baseline map to plan their zoning strategies until more specific and detailed information becomes available.

LCC provides a snap shot of coral reef health as it not only represents the coral dominant reef but also has significant influence on fish species richness and composition where they constitute a positive correlation (Bell and Galzin, 1984). However, LCC cover alone is inadequate as a metric for coral reef health because it does not indicate sufficient information on coral reef resilience (Hughes et al., 2010). The herbivore functional group is becoming increasingly recognized as an important indicator of reef health and resilience. Herbivores play crucial roles in maintaining the natural balance of algae and coral by grazing on macroalgae that competes with corals, preventing a phase shift toward algal dominant reefs and increasing coral reef resilience (Korzen et al., 2011; Bronstein and Loya, 2014). Hence, the keystone algae grazer, Long-spined Sea Urchin and the herbivorous Parrotfish were included as additional components in determining the health of Tioman's coral reefs. On the other hand, DCC suggests the extent of coral reef being damaged due to human and natural disturbances (Hill and Wilkinson, 2004). In this project, DC was an additional substrate category to differentiate natural rocks or rocky shore (RC) from corals that have died due to these disturbances. Natural rock has a smoother surface, has no corallite structure and does not take the typical shapes of corals such as branching, foliose, massive or table. While for DC, the skeletal structure can still be seen albeit slightly eroded compared to RKC. Overall, the combination of all four components of Reef Check indicators (LCC, DCC, LSU, and Parrotfish) were used as proxies to map the coral reefs of Tioman into three distinctive zones with different reef health condition.

Our visual analysis of the zoning map was consistent with and supported by the results from statistical analyses. Our project suggests that Parrotfish was found to be a weak predictor to distinguish different zones and this is likely due to its high mobility. This is expected given that parrotfishes displayed changing home range behavior that is driven by nutritional demand (Welsh et al., 2013). While LSUs are known to be keystone algae grazers, they are also known as bioeroding agents. Hence, too high abundance of LSU can be considered unhealthy to the reefs. Furthermore, higher LSU abundance could also mean that there is proliferation of algae. In this project, the metric of unhealthy LSU abundance is adopted from the study by Carreiro-Silva and Mcclanahan (2001) whereby the abundance of LSU in an unprotected area is approximately 60.00 ind.100 m^{-2} meanwhile the abundance in a protected area is approximately 6 ind.100 $m^{-2.}$

The map produced showed that Tioman can be separated into three distinctive categories of zones and different conservation strategies can be applied to them. Comparatively, based on the visual and statistical analyses, Zone A is considered to be the healthiest, while Zones B and C are less healthy. The different conservation strategies for each zone can be modeled following the Great Barrier Reef Zoning Plan or the categories listed in the IUCN guidelines for protected area management (Great Barrier Reef Marine Park Authority, 2003; Day et al., 2012).

Zone A is proposed to be designated with either Preservation or Scientific Research zone or IUCN Catergory Ia, Ib or even II zone. The objective of this zone should be focused on conserving the ecosystem and associated biodiversity with minimal human impacts. Activities allowed in this zone should be restricted and limited to only research purposes, monitoring, low impact tourism and non-extractive activities. This is to ensure that the many sites of "Good" LCC rating can continue to be preserved without further damage from human activities. There is a local community living within this zone, but it is congregated within a bay. Therefore, some controlled low intensity tourism can be allowed in order to provide some livelihood to the local community. A large part of this zone is mostly rocky formation, uninhabited and exposed to strong wind and waves during monsoon. The mean low abundance of LSU in this zone may suggest low herbivory rate and concomitantly lower bioerosion rate. The low herbivory rate from LSU can be compensated by moderate abundance of Parrotfish to control the proliferation of algae. Bonaldo et al. (2017) found that herbivory rate of parrotfish was 3 to 6 times higher within a MPA and Tioman is already a gazetted MPA with a no-take approach. Excessive growth of algae can inhibit coral settlement, hence lowering coral reef resilience and eventually leading to increased risk of a phase shift toward degraded reef (Bellwood et al., 2004; McManus and Polsenberg, 2004; Hughes et al., 2007). Low LSU abundance also suggest that there is sufficient predation in this area to control LSU's population (Carreiro-Silva and Mcclanahan, 2001) subsequently controlling the bioerosion rate. Therefore, the characteristic of this zone fit to be assigned with the highest conservation priority.

On the other hand, Conservation Park zone or IUCN Category IV zone should be designated for Zone B which comprises several sites mixed with "Good", "Fair," and "Poor" rated LCC, several sites with high DCC (>75%) in addition to numerous sites with high abundance of LSU and moderate abundance of Parrotfish. The primary objectives of this zone should be providing opportunities for sustainable use and recreational activities as well as limited extractive research for academic purposes. The main village of Tioman where all the administration agencies are based is located within this zone. Hence, this zone has reasonable use of resources between stakeholders established for decades for non-extractive recreational activities such as diving and tourism. However, there is a need to control the growing tourism in order to manage the coral reef conditions in a sustainable manner. Our data also highlighted that this zone has high abundance of LSU which may suggest several issues of concern including extraction of their natural predators such as wrasses and triggerfishes, increased level of nutrients and the risk of bioerosion. The high grazing capacity of sea urchins poses a bioerosion threat that can reduce reef stability, growth and resilience (O'Leary and Mcclanahan, 2010). These issues suggest threats of illegal fishing activities and nutrient run-off from tourism operations or land based activities. Hence, Zone B is where most management intervention is recommended.

Zone C is recommended to be designated as General Use zone or IUCN Category VI zone. The objectives of this zone is proposed to be mutually beneficial between conservation and opportunities for sustainable multiple use of the resources. This zone has the poorest coral reef health in terms of LCC and DCC, which also suggest that there is limited livelihood opportunity to the local community residing in this zone. Extractive activities such as traditional, artisanal, recreational fisheries or even research should be allowed. However, there are a few sites with moderate abundance of LSU and Parrotfish to control algal growth. Hence, this zone is highly recommended for habitat rehabilitation or restoration to rebuild the degraded reef and to maintain the balanced population of both herbivorous species.

This project showcases the feasibility of the citizen science approach in mapping baseline information that will be useful for planning of conservation management strategies. The advantage of this approach is that the baseline map can be produced quickly with minimal resources and yet is still sufficiently reliable. Nevertheless, this work was meant to develop a baseline map for management and further studies should be conducted on specific issues carefully when necessary.

AUTHOR'S NOTE

Between the time the work was conducted and the article acceptance, the Department of Marine Park Malaysia, Ministry of Natural Resources and Environment has been restructured and is now parked under the Department of Fisheries, Ministry of Agriculture and Agro-based Industry.

DATA AVAILABILITY

The datasets for this manuscript are not publicly available because this particular work was funded by a governmental agency where access to the data collected from this study must be requested with permission. Requests to access the datasets should be directed to the corresponding author.

AUTHOR CONTRIBUTIONS

CL, AK-A, YA, JH, AL, and NZ contributed the conception and design of the work. AK-A, AC, YSL, YLL, PM, VL, AM, YM, and RM conducted the data collection. CL performed the statistical analysis and GIS. CL and AK-A wrote the manuscript. All authors read and approved the submitted version.

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Caribbean Near-Shore Coral Reef Benthic Community Response to Changes on Sedimentation Dynamics and Environmental Conditions

*Abimarie Otaño-Cruz[1,2,3], Alfredo A. Montañez-Acuña[2,3,4], Noelia M. García-Rodríguez[2,3,5], Dakeishla M. Díaz-Morales[3,6], Elizabeth Benson[7], Elvira Cuevas[2,8], Jorge Ortiz-Zayas[1,2] and Edwin A. Hernández-Delgado[1,2,3]**

[1] Department of Environmental Sciences, University of Puerto Rico, San Juan, Puerto Rico, [2] Center for Applied Tropical Ecology and Conservation, University of Puerto Rico, San Juan, Puerto Rico, [3] Sociedad Ambiente Marino, San Juan, Puerto Rico, [4] Graduate School of Planning, University of Puerto Rico, San Juan, Puerto Rico, [5] Graduate School of Public Health, University of Puerto Rico, San Juan, Puerto Rico, [6] Department of Aquatic Ecology, University of Duisburg-Essen, Duisburg, Germany, [7] Faculty of Arts and Sciences, Harvard University, Cambridge, MA, United States, [8] Department of Biology, University of Puerto Rico, San Juan, Puerto Rico

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***Correspondence:**
Edwin A. Hernández-Delgado
edwin.hernandezdelgado@gmail.com

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Coral reefs are facing unprecedented global, regional and local threats that continue to degrade near-shore habitats. Water quality degradation, due to unsustainable development practices at coastal watersheds, is one of the greatest stressors across multiple spatial scales. The goal of this study was to assess near-shore coral reef benthic community spatio-temporal response to sedimentation patterns, weather, and oceanographic dynamics at Bahía Tamarindo and Punta Soldado in Culebra Island, Puerto Rico. Benthic data were collected across a distance gradient from the shore through high-resolution images at marked belt transects. Environmental data were assessed and contrasted with benthic assemblages using multivariate correlations and multiple linear regression. Coral colony abundance and coral recruit assemblages showed significant variation among seasons, sites and distance zones (PERMANOVA, $p < 0.01$). Species diversity (H'n) increased at both study sites with distance from shore, and the most conspicuous coral recruit species were stress-tolerant *Porites astreoides*, *P. porites,* and *Siderastrea radians*. Difference in coral abundance and coral recruits per site had a strong significant negative relationship with sediment characteristics and depth ($p < 0.05$). Near-shore coral reef benthic community structure was significantly different between sites and distance zones from shore, with depth having an important role in shaping reef zonation. Changes in benthic community structure were associated with local sediment distribution patterns emerging from human alteration of coastal watersheds and natural events that cause terrigenous sediment deposition and sand resuspension across the reef. Coral cover was significantly lower at zones more exposed to recurrent sedimentation stress ($p < 0.01$). It was also correlated with sediment texture ($p = 0.006$) and terrigenous sediment deposition ($p = 0.016$). Scleractinian coral

cover had an inverse relationship with gorgonian and macroalgae cover. In a short-term period, a pattern of increased dominance of encrusting calcareous algae *Ramicrusta textilis* and invasive sponge *Dictyonella funicularis* were documented. Changing land use and increased frequency of extreme weather events, as a consequence of global patterns of climate change, may play an important role shaping near-shore coral reefs benthic communities and could threaten the resilience of coastal regions. Therefore, collaborative and trans-disciplinary ecosystem-based management efforts are urgently needed to effectively reduce land-based stressors and foster near-shore coral reef recovery.

Keywords: benthic cover, coral abundance, coral diversity, coral recruit, coral reef community structure, environmental stressors, near-shore coral reefs, sedimentation dynamics

INTRODUCTION

Coral reefs are affected by a wide array of global, regional, and local stressors that have led to habitat degradation worldwide during recent decades (Hughes, 1994; Gardner et al., 2003; Wilkinson and Souter, 2008). Global and regional trends of coral decline are associated to anthropogenic stressors combined with climate change-related impacts and natural disturbances (Eakin et al., 2010; Wild et al., 2011; Bozec and Mumby, 2015; Hernández-Delgado, 2015). At a local scale, land-based stressors represent a major threat to near-shore coral reefs in the Caribbean region. These land-based stressors are mostly related to increasing trends of land-use changes, coastal urban sprawl and tourism-based activities with direct effects on sediment-laden runoff and sediment distribution along coastal waters (Rogers, 1990; Larsen and Webb, 2009; Hernández-Delgado et al., 2012; Ramos-Scharrón et al., 2012, 2015; Bégin et al., 2013). This is particularly true for small tropical semi-arid islands, where an alteration of sediment delivery and distribution dynamics along near-shore reef ecosystem has been documented as a result of watershed alteration combined with changes in local weather patterns (Otaño-Cruz et al., 2017). Increased terrigenous sediment influx to coastal waters increases coral reef vulnerability and susceptibility to experience phase-shift toward alternate states, often dominated by non-reef building taxa and macroalgal assemblages (Acevedo et al., 1989; Fabricius, 2005, 2011; Bellwood and Fulton, 2008; Dudgeon et al., 2010; Hughes et al., 2010). This combination of factors can have profound permanent effects on reef ecosystem functions and services (Acevedo et al., 1989; Bellwood et al., 2004; Hughes et al., 2010). Therefore, there is a major concern regarding the potential effects of elevated terrestrial sediment input to coastal waters and changes in environmental conditions, especially on coral reef habitats historically adapted to low sedimentation levels.

The coral reef response to local human-induced disturbances depends on the frequency, duration, and distance from the source of the environmental stressor (Fabricius, 2005; Smith et al., 2008; Edmunds and Gray, 2014). Recurrent environmental disturbances have adverse implications for coral reef benthic communities by producing shifts in ecological dynamics and in the population of endangered coral species (Díaz-Ortega and Hernández-Delgado, 2014). Sedimentation stress has been associated with localized partial coral mortality, reduced coral growth rate, inhibited larval settlement, and reduced fish grazing (Loya, 1976; Nugues and Roberts, 2003a,b; Fabricius, 2005; Bellwood and Fulton, 2008). Impacted coral reefs can have loses of sensitive species, thus reinforcing phase shifts toward sediment-resistant species, and dominance by algae and other non-reef building taxa (Acevedo et al., 1989; Bellwood et al., 2004; Fabricius, 2005, 2011; Hughes et al., 2010). Changes in species composition can produce significant changes in structural complexity and functioning by reducing reef accretion and rugosity (Alvarez-Filip et al., 2013). Coral reefs exposed to recurrent sediment pulses and high turbidity levels also have limited ability to recover after chronic disturbances [e.g., bleaching caused by high sea surface temperature (SST)] and increased prevalence of coral diseases (Cróquer et al., 2002; Toledo-Hernández et al., 2007; Pollock et al., 2014; Stubler et al., 2016). Therefore, the understanding of the interactions between land and sea ecosystems and the short-term response of coral reefs community to sedimentation stressors is paramount for the development and implementation of effective and adaptive ecosystem-based management strategies to prevent further decline in live coral cover and enhance coral reef resilience (Rivera-Monroy et al., 2004; Alvarez-Filip et al., 2009; Mumby and Steneck, 2011).

Live coral cover decline across the Caribbean region has been partly attributed to increased sediment delivery to coastal waters after storms and heavy rainfall events, as a consequence of coastal watershed alteration and unsustainable development trends (Ramos-Scharrón and MacDonald, 2007; Hernández-Delgado et al., 2011, 2012, 2014b; Ramos-Scharrón et al., 2012; Sturm et al., 2014). At a broader spatial scale, the combined effects of chronic anthropogenic stressor impacts, such as declining water quality due to increased land-based source of pollution (LBSP), reduction in grazing due to overfishing and changing climate (e.g., change in rainfall patterns), have contributed to near-shore coral reef ecosystem degradation (Miller et al., 2009; Hernández-Pacheco et al., 2011; Edmunds, 2013; Jackson et al., 2014). This phenomenon illustrates the complexity and interconnectedness between the coastal and marine habitats, and thus, represents unfavorable conditions for coral reef recovery (Hughes and Connell, 1999; Rogers and Miller, 2006; Ennis et al., 2016). The implementation of mitigation and restoration

projects at watershed and coral reef scales has become a management priority and the understanding of the existing land-sea and climate interconnectedness becomes critical for the implementation of rapid and effective strategies to contribute to coral reef recovery worldwide.

The goal of this study was to assess whether there was a significant spatio-temporal difference in the short-term response of benthic communities to variations in sedimentation patterns and environmental variables. Therefore, this study aimed to: (i) assess variation in coral reef benthic community structure through coral colony abundance, coral recruit abundance, percent live coral cover, octocoral, sponge and macroalgae cover in a distance gradient from the shore; and (ii) contrast spatio-temporal changes in coral colony abundance, coral recruit abundance, and coral cover with sedimentation patterns, and environmental variable dynamics.

MATERIALS AND METHODS

Data Acquisition, Field Sampling, and Laboratory Analyses

Culebra Island is a mid-shelf semi-arid island located 27 km off the eastern coast of Puerto Rico, in the northeastern Caribbean Sea. The study was conducted from February 2014 to April 2015 across two leeward coral reef locations: Bahía Tamarindo (BTA, 18°18′ N, 65°19′ W) and Punta Soldado (PSO, 18°16′N, 65°17′W) (**Figure 1**). Both study sites are part of a long-term community-based coral reef rehabilitation effort to support reef functions and services, including the reefs' role as fish nursery grounds (Hernández-Delgado et al., 2018). Transplant of *Acropora* spp. onto the reef was conducted prior to the initiation of this research in 2003. Belt transects of 10 m^2 were marked parallel to the coastline and were assessed with high-resolution replicate photo-quadrat every 1 m^2. Marked transects were assessed seasonally, from spring to winter, every 3 months, during a 1 year period. Each monitoring station consisted of triplicate fixed transects. Benthic assessment in BTA was conducted across a total of three replicate monitoring stations within distance zone A (<60 m from shore), with a depth range from 1 to 2 m, and three stations in distance zone B (>60 m from shore), with depth range contour from 2 to 4 m. In PSO, there were a total of two monitoring stations within zone A and B and the depth range contour was similar to BTA.

Scleractinian coral, hydrocoral, and octocoral species assemblages were assessed within each transect using photo-quadrats to identify corals to the lowest taxonomic level possible and to calculate the average abundance for each sampling station. This data was used to calculate coral colony abundance, species richness (S), species diversity index (H'n) (Shannon and Weaver, 1948), and evenness index (J'n) (Pielou, 1966) at each site and distance zone. Scleractinian coral recruit abundance was also assessed, specifically for colonies ≤4 cm in diameter for larger species (i.e., *Pseudodiploria* spp., *Siderastrea siderea*), and ≤2 cm for smaller species (i.e., *Porites astreoides*) (Dueñas et al., 2010). The benthic cover was assessed from high-resolution images by digitally projecting 48 regularly-distributed dots over each photo-quadrat image. Benthic components under each dot were identified, including scleractinians, hydrocorals, octocorals, sponges, macroalgae, algal turf, coralline algae (CA), and cyanobacteria, among other components (i.e., sand, pavement, rubble). Coral cover as recorded included both Scleractinians and hydrocorals. Algal turf was classified as a mix of short algae (<1 cm) and sediments (NOAA, 2015). Data was used to calculate percent benthic component cover and benthic community structure.

The environmental database included sedimentation rate, sediment texture, and composition. Sediment traps were deployed at both sites across a distance gradient from shore, within a distance of less than 10 m from marked belt transects. Sediment samples were collected on a monthly basis and processed at the laboratory. Sediment samples were oven-dried at 60°C for 24 h and total dry weight was recorded to calculate sediment accumulation rate (Edmunds and Gray, 2014; Otaño-Cruz et al., 2017). Sediment texture (silt-clay <63 μm and sand >63 μm) were analyzed through dry sieving, while loss on ignition techniques were applied to determine the organic matter (3 h oven dry 550°C), carbonate matter (3 h oven dry 950°C) and terrigenous sediment composition (Otaño-Cruz et al., 2017). Rainfall events were documented with HOBO RG3 (Onset Computers, Co.) rain gauges located in coastal watersheds near study sites, while sea surface temperature at both study sites were recorded with HOBO Watertemp Pro V2. Oceanographic hydrodynamics data, such as wave height and wind speed, were acquired from the Caribbean Integrated Coastal Ocean Observing System (CariCOOS) online database[1], buoy NDBC 41056 located at Vieques sound.

Statistical Analyses

Benthic components were tested using four-way non-parametric permutational analysis of variance (PERMANOVA) and pairwise comparison for the fixed factors of seasons, time, site and distance from shore (Anderson et al., 2008). Multivariate analyses were performed in Primer v7 + PERMANOVA v1.16 software (Quest Research Ltd., Auckland, New Zealand) to analyze spatio-temporal variation in coral colony abundance, coral recruit abundance, and benthic community structure (Clarke et al., 2014; Clarke and Gorley, 2015). A significant relationship was identified by factors that had $P < 0.05$. The rank order of dissimilarities were calculated through Bray-Curtis resemblance from coral abundance, coral recruit abundance, and benthic community matrices. For coral recruitment, a zero-adjusted Bray-Curtis resemblance matrix was calculated, including a "dummy variable" to reduce distortion from absent species by samples (Clarke et al., 2006). Coral colony abundance, coral recruit abundance, and percent benthic community parameters were standardized to balance the contribution of common and rare species, and thus, represented the relative percentage of species for each sample. Species assemblages were also square root-transformed prior to analysis to meet assumptions of normality and homogeneity of variance (Clarke and Warwick, 2001; Gotelli and Ellison, 2013). All multivariate tests were

[1]http://www.caricoos.org/data-download

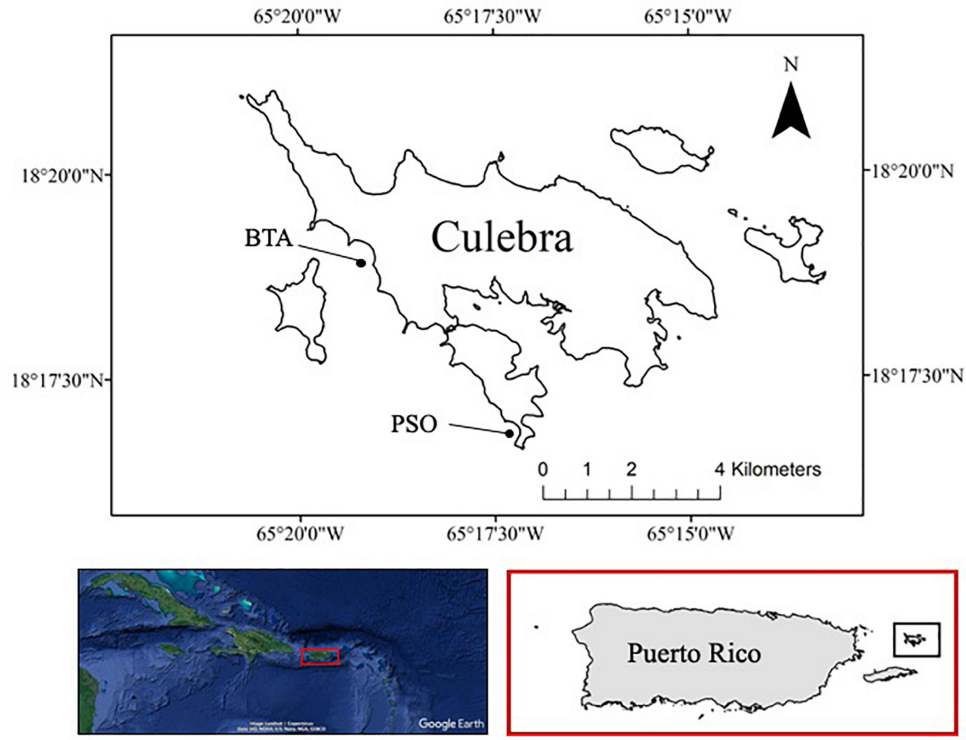


FIGURE 1 | Study site at Culebra Island, Puerto Rico. Bahía Tamarindo (BTA) and Punta Soldado (PSO).

based in 10,000 permutations (Hernández-Delgado et al., 2014a). Sigma Plot v.11 (Systat Software, Inc.) was used for graphical representation of biological data.

Ordination was performed using non-metric multi-dimensional scaling (nMDS) and principal coordinates ordination (PCO), by calculating the distance among centroids, to display the variations in benthic communities in a three-dimensional space and determine which benthic component explained spatio-temporal variation. Cluster and similarity profile test (SIMPROF) were used to identify groups with similarity differences between samples, to test the null hypothesis of no significant spatial and temporal differences on multivariate structure of benthic assemblages. Afterward, a similarity percentage (SIMPER) analysis routine was performed to determine which key taxa contributed most to similarities and spatial variation in the benthic community structure through time, within sites and distance (Clarke et al., 2014).

Environmental variables were correlated with the biological matrices using non-parametric multivariate correlation routine BEST-BIO ENV (Spearman rank correlation) to determine the best environmental variable that explained differences in coral reef benthic community spatio-temporal variation (Clarke et al., 2014). RELATE routine was used to test the relationship between coral recruit abundance and percentage macroalgal cover, with sedimentation and other abiotic variables. DISTLM was performed for multiple linear regression analysis to assess potential effects of environmental variables on coral colony abundance, coral recruit abundance, and benthic cover with a distance-based redundancy analysis (dbRDA) (Anderson et al., 2008).

RESULTS

Coral Colony Abundance, Coral Species Richness, and Diversity

Mean coral colony abundance (±95% confidence interval) at Bahía Tamarindo and Punta Soldado near-shore coral reefs was 13.18 ± 0.92 m^{-2}. There were significant differences in coral abundance by season (Pseudo $F = 2.38$, $p = 0.0002$), sites (Pseudo $F = 36.96$, $p = 0.0001$), and distance zones (Pseudo $F = 9.98$, $p = 0.0001$) (**Table 1**). Coral colony abundance was higher at BTA than PSO with a mean of 13.77 ± 0.92 m^{-2}. The highest coral abundance value at BTA was documented a long distance zone B (farther from shore, >60 m from shore), with a mean of 17.95 ± 1.27 m^{-2} (**Figure 2**). The lowest value of coral abundance at BTA was recorded at distance zone A (near to shore, <60 m from shore) with a mean of 9.59 ± 0.94 m^{-2}. Mean coral abundance was lowest at PSO with a mean 12.29 ± 1.29 m^{-2}. The highest coral colony abundance value at PSO was documented near shore, zone A, with a mean of 12.78 ± 4.68 m^{-2}. Coral abundance at both sites experienced significant variation through seasons at offshore zone (**Figure 2**). There were significant interaction effects between site and distance zones (Pseudo $F = 18.74$, $p = 0.0001$), suggesting the influence of the latter on the observed variation between sites. Pairwise analysis indicated that

TABLE 1 | Results of permutational analysis of variance (PERMANOVA) for coral colony abundance and coral recruit abundance.

Factors	df	Pseudo *F*	*P* (perm)
Coral colony abundance			
Season (Se)	3	1.71	**0.0112**
Site (Si)	1	32.09	**0.0001**
Distance (Di)	1	10.16	**0.0001**
Se × Si	3	1.35	0.0933
Se × Di	3	0.77	0.8141
Si × Di	1	18.74	**0.0001**
Se × Si × Di	3	0.75	0.8338
Coral recruit abundance			
Season (Se)	3	2.72	**0.0030**
Site (Si)	1	27.30	**0.0001**
Distance (Di)	1	8.00	**0.0001**
Se × Si	3	2.25	**0.0100**
Se × Di	3	1.47	0.1500
Si × Di	1	11.29	**0.0001**
Se × Si × Di	3	1.60	0.1100

Bold p-values represent significance.

coral colony abundance had significant variation in BTA ($t = 4.02$, $p = 0.0001$) and PSO ($t = 4.02$, $p = 0.0001$) by distance zones.

A total of 66 species were documented (34 scleractinians, 2 hydrocorals, and 30 octocorals) in both study sites combined. The scleractinian coral species with highest mean relative abundance at BTA were *Porites astreoides* (24.54%), *P. porites* (11.40%), *Acropora cervicornis* (5.84%), *Siderastrea siderea* (5.76%), *Pseudodiploria strigosa* (5.66%), *Diploria labyrinthiformis* (5.03%), *S. radians* (4.59%), and *Agaricia agaricites* (3.48%), representing a total 66.30% of species contribution (**Figure 3**). The only species that showed a pattern of increased relative abundance through time at both distance zones was *P. porites.* The hydrocorals *Millepora alcicornis* and *M. complanata* had a higher percentage relative abundance at BTA than PSO, with a mean of 16.54 and 2.35%, respectively. Coral species with the highest mean relative abundance at PSO were *P. astreoides* (31.06%), *P. porites* (10.15%), *O. annularis* (10.14%), *S. siderea* (3.30%), and *P. strigosa* (2.53%), representing a total of 57.18%. Relative abundance *D. labyrinthiformis* and *P. porites* decreased through seasons near to shore and increased farther from shore. The coral species *P. astreoides* and *O. annularis* had higher relative abundance at reef zones farther from shore.

Gorgonians had a higher relative abundance at PSO than BTA, with a mean of 11.87%, mainly composed by *Antillogorgia americana*, *Eunicea flexuosa, Gorgonia ventalina*, and *Plexaura homomalla* (**Figure 3**). In contrast, gorgonians at BTA represented a mean relative abundance of 5.08%. The species that showed patterns of increased relative abundance through time at PSO at both distance zones were *A. cervicornis, A. agaricites,* and the octocoral *P. homomalla.* The SIMPER analysis revealed that the species that mostly contributed to differentiate coral colony abundance between sites were *O. annularis* (7.45% contribution), *M. alcicornis* (6.41%), *P. strigosa* (4.86%), and *S. radians* (4.83%), with a total of 23.55% of the observed variation. Differences between distance zones were mostly attributed to *M. alcicornis* (7.31%), *P. porites* (5.82%), *P. astreoides* (5.44%), and *O. annularis* (4.92%), with a total of 23.49% of the observed variation. PCO analysis identified four major groups from the coral abundance structure that represent the interaction between sites and distance zones (**Figure 4**). The seasonal variation of coral community within distance zone A had higher similarity between BTA and PSO due to the proximity of both clusters. In contrast, the coral community at PSO distance zone B showed a major difference between sites and distances, with clusters farther apart. The proposed PCO explains 74% of the variation between and within groups.

Coral species richness (S) at BTA was highest at distance zone B with a mean of 18.11 ± 1.18 ($\pm$CI 95%). Mean S decreased from spring to winter at both distance zones (**Figure 5A**). Overall, the highest total coral species richness between sites was recorded at PSO distance zone B with a mean of 18.63 ± 1.19. Species richness at PSO experienced seasonal variation along both distance zones (**Figure 5B**). Species richness showed significant differences by distance zones (Pseudo $F = 1.37$, $p = 0.0010$), but no difference was recorded between seasons (Pseudo $F = 1.41$, $p = 0.2300$), or sites (Pseudo $F = 0.91$, $p = 0.3300$).

Coral species diversity (H'n) showed contrasting patterns by sites. In BTA, H'n was higher farther from shore, with the highest values recorded during the summer with a mean of 2.30 ± 0.14 (**Figure 5C**). At both distance zones, H'n declined gradually from spring to winter season, representing a short-term percent change of -5.98%. The highest H'n between sites was recorded at PSO farther from shore with a mean of 2.27 ± 0.07. At PSO, H'n increased from spring to winter at both distance zones (**Figure 5D**). Coral diversity at both sites had lower values at shallower, near-to-shore areas. There were significant differences by distance zones (Pseudo $F = 32.22$, $p = 0.0010$), but no difference was recorded between seasons (Pseudo $F = 0.44$, $p = 0.7200$) or sites (Pseudo $F = 0.04$, $p = 0.8500$). The evenness (J'n) showed that BTA had a higher difference in species dominance between distance zones, with a higher evenness farther from shore (**Figure 5E**). J'n at BTA showed a pattern of rapid decline at distance zone A through seasons, reflecting greater changes in species dominance. J'n at PSO had similar patterns of increasing evenness across both distance zones through seasons (**Figure 5F**). There was a significant difference in J'n by distance zones (Pseudo $F = 12.24$, $p = 0.0020$).

Coral Recruit Abundance

Coral recruit community structure showed statistically significant differences among seasons (Pseudo $F = 2.72$, $p = 0.0030$), site (Pseudo $F = 23.30$, $p = 0.0001$), and distance (Pseudo $F = 8.00$, $p = 0.0001$) (**Table 1**). There were also significant differences between the interactions of site and distance (Pseudo $F = 11.29$, $p = 0.0001$). Coral recruit abundance was higher in BTA at areas farther from shore with a mean of 1.96 ± 0.46 m^{-2}. Mean coral recruit abundance at BTA increased from 0.88 m^{-2} in spring to 2.96 m^{-2} in winter. Coral recruit abundance near to shore increased from 0.28 (spring) to 1.28 m^{-2} (winter), while farther from shore increased from 1.47 to 4.63 m^{-2} (**Figure 2**). In contrast, at PSO, coral recruit abundance was higher on reef

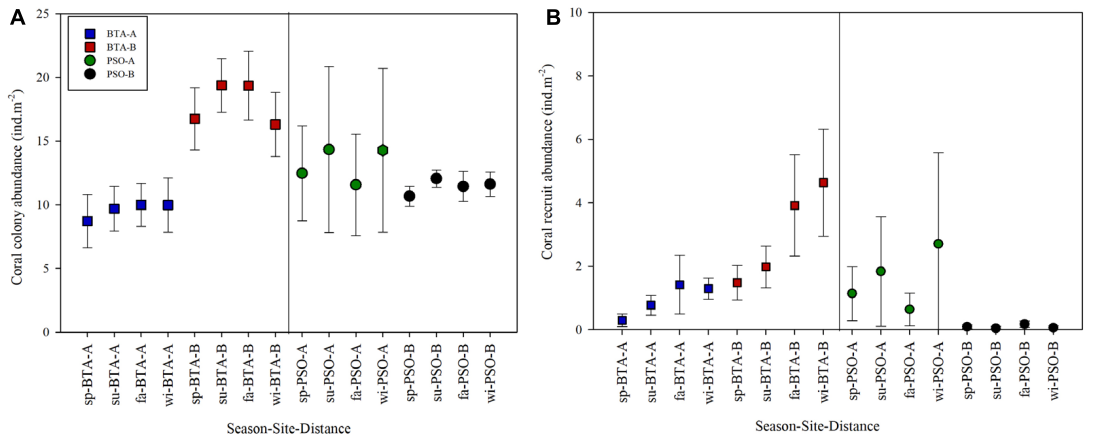


FIGURE 2 | Mean coral colony abundance (mean ± 95% CI) **(A)** and coral recruit abundance **(B)** by season, site, and distance zone. Coral abundance includes hard corals, hydrozoans and gorgonians. Blue and red squares represent Bahía Tamarindo (BTA); green and black circles represent Punta Soldado (PSO). Seasons are defined as: sp, spring; su, summer; fa, fall; wi, winter.

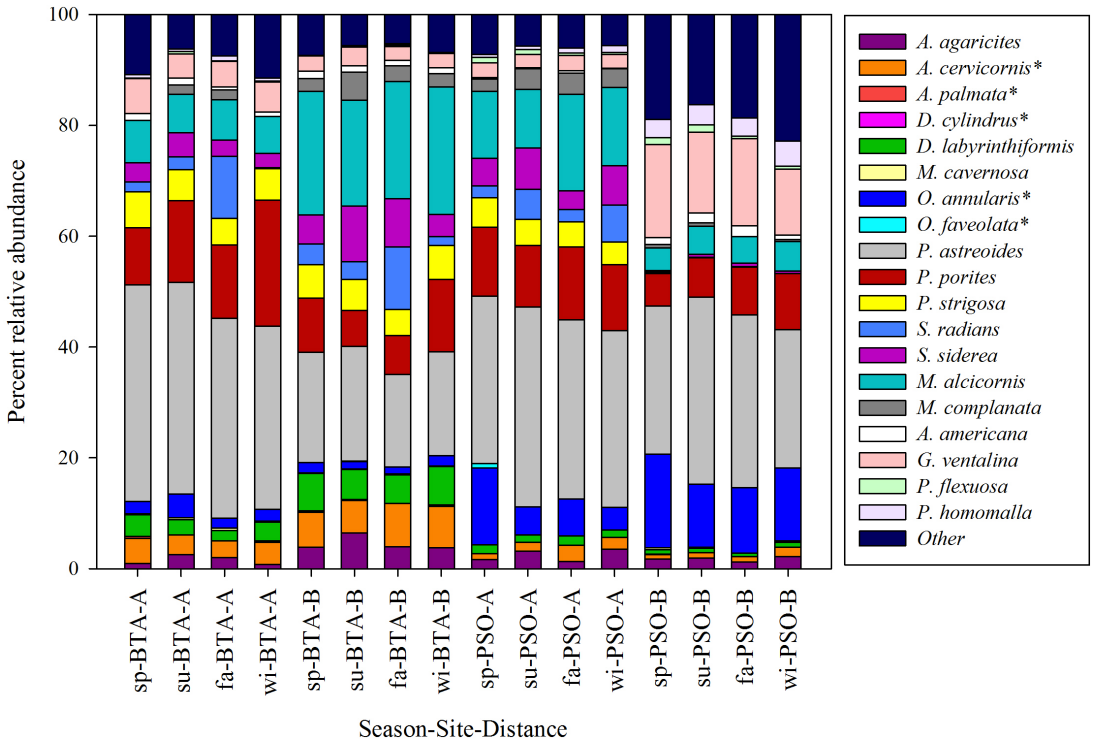


FIGURE 3 | Percent relative abundance of scleractinians, octocorals and octocoral species by season, site and distance zones. Colors represent coral species and asterisk identifies coral species listed as threatened under the Endangered Species Act (Federal Register, 2014). Seasons are defined as: sp, spring; su, summer; fa, fall; wi, winter.

areas closer to shore. Mean recruit abundance increased from 0.60 m^{-2} in spring to 1.38 m^{-2} in winter. Recruit abundance near to shore increased from 1.13 to 2.7 m^{-2} and farther from shore decreased gradually from 0.08 to 0.05 m^{-2}. A total of 11 coral species recruited at both study sites and the species with highest percentage relative abundance were *Siderastrea radians* (45.1%), *S. siderea* (23.1%), and followed by *P. astreoides* (19.2%) (**Table 2**). No coral recruits of large reef-building coral species were documented at BTA through the study period.

PCO analysis identified two major groups which distinguished different coral recruit abundance patterns of PSO distance zone B from zone A, and from BTA among all seasons (**Figure 6**). There was also one outlier event identified for PSO zone B during the summer season, particularly due to a

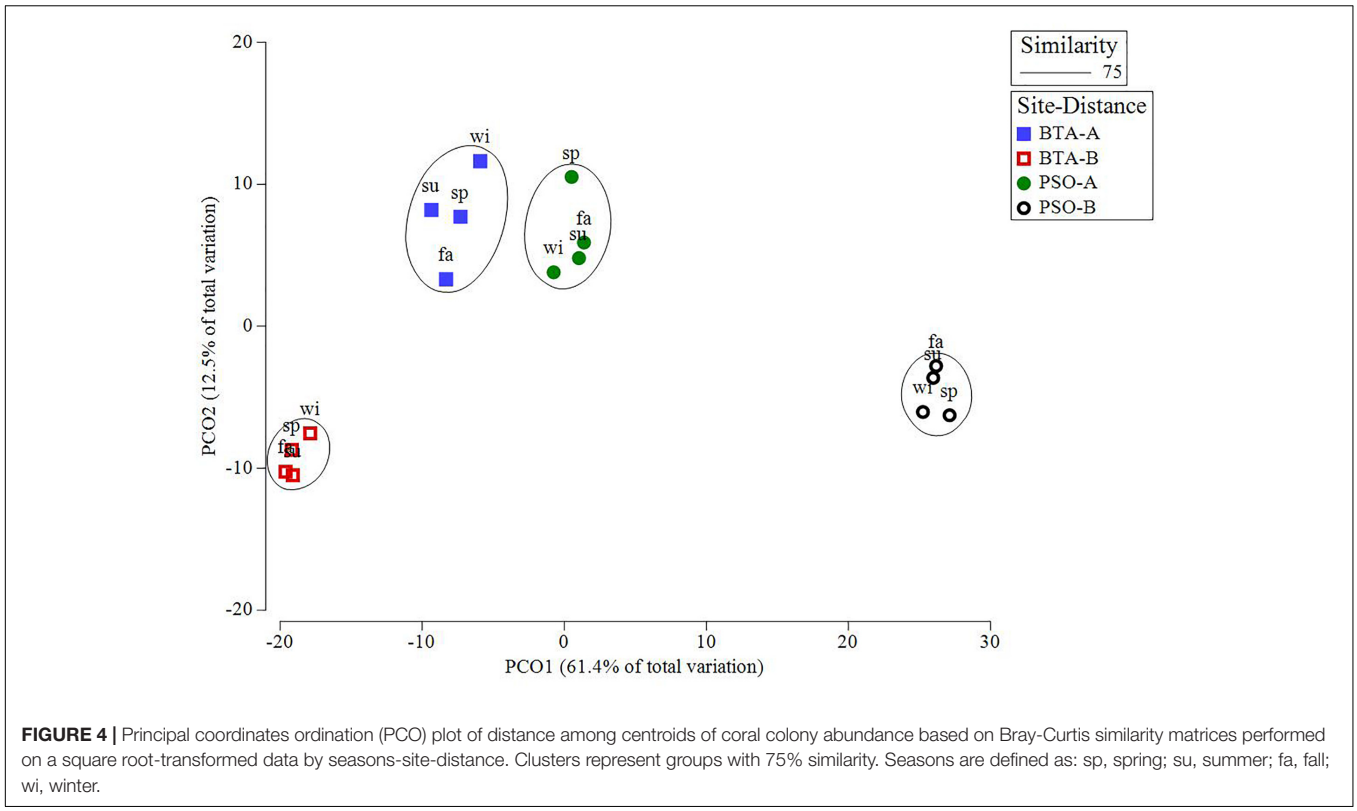


FIGURE 4 | Principal coordinates ordination (PCO) plot of distance among centroids of coral colony abundance based on Bray-Curtis similarity matrices performed on a square root-transformed data by seasons-site-distance. Clusters represent groups with 75% similarity. Seasons are defined as: sp, spring; su, summer; fa, fall; wi, winter.

significant reduction recorded with a mean recruit abundance of 0.4 m^{-2}. However, SIMPROF analysis only identified two distinct groups for coral recruit abundance. The calculated PCO explains 83.2% of the total variation. Also, SIMPER analysis showed the three species that contributed to explain 79% of the differences of coral recruit abundance patterns between sites were *S. radians* (32.41% individual contribution), *P. astreoides* (23.93%), and *S. siderea* (22.42%). The species that contributed to explain 73% of the differences between distance zones were *P. astreoides* (27.83%), *S. siderea* (24.42%), and *S. radians* (21.21%). The average dissimilarity was 49.69 between sites and 61.01 between distance zones.

Benthic Components Percent Cover and Community Structure

Live coral cover was significantly higher on BTA zone B with a mean percent cover of 17.87 ± 0.67%, and a gradual reduction was recorded in a short-term period (**Figure 7A**). Percent coral cover at zone A, near to shore, remained relatively stable through seasons. Coral cover at PSO had the lowest percentage cover farther from shore with a mean of 7.84% and it slightly decreased through time at both distance zones. Percentage coral cover was significantly different between sites (Pseudo F = 25.26, p = 0.0010), distance (Pseudo F = 13.77, p = 0.0010), and the interaction site by distance (Pseudo F = 18.82, p = 0.001) (**Table 3**).

Stress-tolerant coral species, *P. astreoides* and *P. porites,* had higher percent relative cover in BTA zone A than distance zone B, and PSO, with a mean of 4.28 and 2.71%, respectively. The hydrocoral *M. alcicornis* had a higher percent cover farther from shore (**Figure 8**). Percentage cover of *A. cervicornis* was higher in distance zone B with a mean cover of 1.95% and increased from spring 1.56% to winter 2.35%. In PSO, the species with higher percentage cover near to shore was *P. astreoides* with a mean of 2.51% and *O. annularis* farther from shore with a mean of 3.63%. The similarity percentage (SIMPER) analysis revealed that 8 species constituted 75% of the observed variations in percent coral cover between sites and distance zones. The species that had major contributions to variations between sites and distance zones were *M. alcicornis*, *O. annularis*, *P. porites*, and *A. cervicornis*. There was an average dissimilarity between sites of 57.11 and 49.85% between distances zones.

Octocorals mostly dominated benthic cover at PSO farther from shore with a mean of 8.10% (**Figure 7B**). The octocoral cover near to shore had similar patterns at PSO and BTA with an overall mean cover of 1.80%. Sponges had higher percentage cover at BTA with a mean of 5.77%. Sponges had a pattern of increasing percentage cover at areas farther from shore, reaching benthic cover of up to 12% during the fall and winter season (**Figure 7C**). The gray encrusting and invading sponge species *Dictyonella funicularis* had a significant contribution at BTA farther from shore, with an increasing benthic cover from spring (0.02%) to fall (7.81%). This species was documented overgrowing dead or diseased *A. cervicornis* colonies, among other benthic components. In contrast, *D. funicularis* had higher percent cover at PSO closer to shore, reef area that experienced an increase in sponge cover from spring to winter.

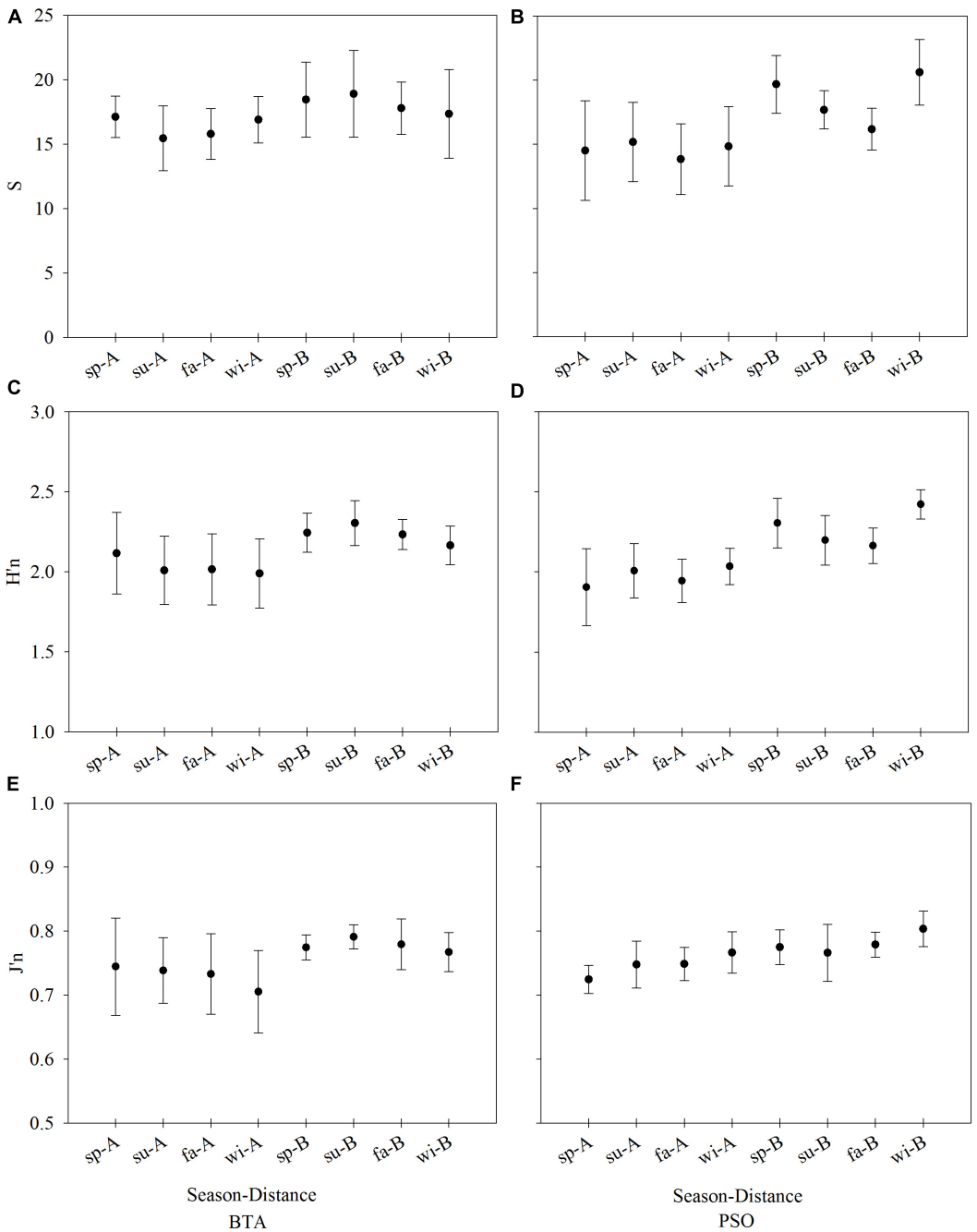


FIGURE 5 | Coral species richness (S, **A,B**), diversity index (H'n, **C,D**) and evenness (J'n, **E,F**) by site, season, and distance zones (mean ± 95% CI). Left column graphs represent Bahía Tamarindo (BTA) and right column Punta Soldado (PSO). Seasons are defined as: sp, spring; su, summer; fa, fall; wi, winter.

Octocoral and sponge cover showed a significant difference among seasons, sites, and distance zones ($p < 0.05$) (**Table 3**). Sponge community also had a significant interaction between site and distance ($p < 0.05$).

Macroalgae and algal turf were important components of the community composition. Macroalgal assemblages had significant differences among season, sites, distance and the interaction site by distance ($p < 0.05$) (**Table 3**). Higher percent macroalgae cover was recorded at PSO, especially at areas farther from shore, where it reached up to 35% (**Figure 9A**). Macroalgae dominance at PSO occurred while the encrusting and invasive red algae *Ramicrusta textilis* overgrew dead skeletons of the *O. annularis* species complex. Reef zones adjacent to the shoreline were dominated by *Dictyota* spp. algae, which reached its highest cover at BTA with 14%. Algae composition at BTA also had a high dominance of turf, which constituted a mean cover of 37.51%, and it was

TABLE 2 | Results of permutational analysis of variance (PERMANOVA) for benthic components by factors.

Factors		Coral		Octocoral		Sponge		Macroalgae		Benthic community structure	
	df	Pseudo *F*	*P*	Pseudo *F*	*P*	Pseudo *F*	*P*	Pseudo *F*	*P*	Pseudo *F*	*P*
Season	3	0.59	0.9102	2.24	**0.0011**	1.53	**0.0396**	2.69	**0.006**	1.96	**0.0021**
Site	1	25.26	**0.0001**	10.43	**0.0001**	9.68	**0.0001**	20.26	**0.0001**	21.00	**0.0001**
Distance	1	6.91	**0.0001**	6.50	**0.0001**	7.33	**0.0001**	6.85	**0.0007**	13.77	**0.0001**
Se × Si	3	0.76	0.7444	1.03	0.4286	1.02	0.4317	1.27	0.2553	1.95	**0.0019**
Se × Di	3	0.28	0.9978	0.72	0.8342	1.04	0.3902	1.82	**0.0707**	0.65	0.9416
Si × Di	1	16.88	**0.0001**	5.54	**0.0001**	9.49	**0.0001**	12.14	**0.0001**	18.82	**0.0001**
Se × Si × Di	3	0.68	0.8337	0.68	0.8712	1.80	**0.0048**	2.54	**0.0091**	1.84	**0.0048**

Bold p-values represent significance.

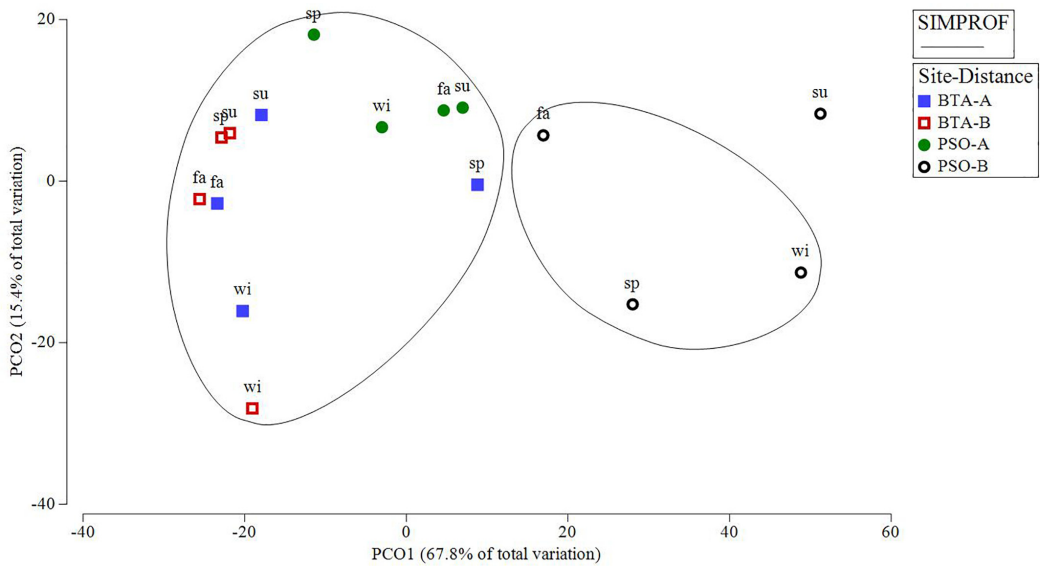


FIGURE 6 | PCO plot of distance among centroids of coral recruit abundance based on Bray-Curtis similarity matrices performed on square root-transformed data by seasons-site-distance clusters. Clusters represent significant SIMPROF groups (~75% similarity within groups). This model explained 83.2% of the observed spatio-temporal variation in coral recruit abundance by site and distance zone. Blue and red squares represent Bahía Tamarindo (BTA); green and black represent Punta Soldado (PSO). Seasons are defined as: sp, spring; su, summer; fa, fall; wi, winter.

significantly higher at zone A with its highest level recorded during the fall season of 51.67% (**Figure 9B**). Meanwhile, on distance zone B, the most prominent algae cover were turf and crustose coralline algae (CCA), *Porolithon* spp., since these constituted a mean cover of 28.66 and 9.53%, respectively. The percentage cover of CCA at BTA farther from shore increased through seasons (**Figure 9C**). Cyanobacterial cover also had an important influence on BTA zone B with increased percentage cover form spring (3.91%) to summer (6.99%) (**Figure 9D**).

Coral reef benthic community structure presented a statistically significant difference among seasons (Pseudo F = 1.96, p = 0.0020), sites (Pseudo F = 21.00, p = 0.0010), distance (Pseudo F = 13.07, p = 0.0010), and within the interactions season by site (Pseudo F = 1.95, p = 0.0070), and site by distance (Pseudo F = 18.82, p = 0.0010) (**Table 3**). All of the benthic components analyzed in this study showed significant differences in the interaction of site by distance. The PCO analysis identified four major groups that distinguished benthic community structure between sites and distance zone (**Figure 10**). Benthic communities were similar between sites at areas near to shore and greater differences between benthic communities were identified farther from shore.

Benthic Community Structure and Environmental Variables

Among the environmental variables assessed, coral colony abundance had a significant negative correlation with sand (>63 μm) (RELATE, Rho = 0.475, p = 0.002), silt-clay (<63 μm) (Rho = 0.422, p = 0.003), carbonate (Rho = 0.263, p = 0.026), and terrigenous sediment content (Rho = 0.328, p = 0.009) (**Table 3**). Coral colony abundance had a significant positive interaction with depth (Rho = 0.858, p = 0.037) at BTA and a negative interaction at PSO. The nMDS bubble plot showed that sand and terrigenous sediment had a significant negative association with spatial configuration of coral colony abundance at BTA,

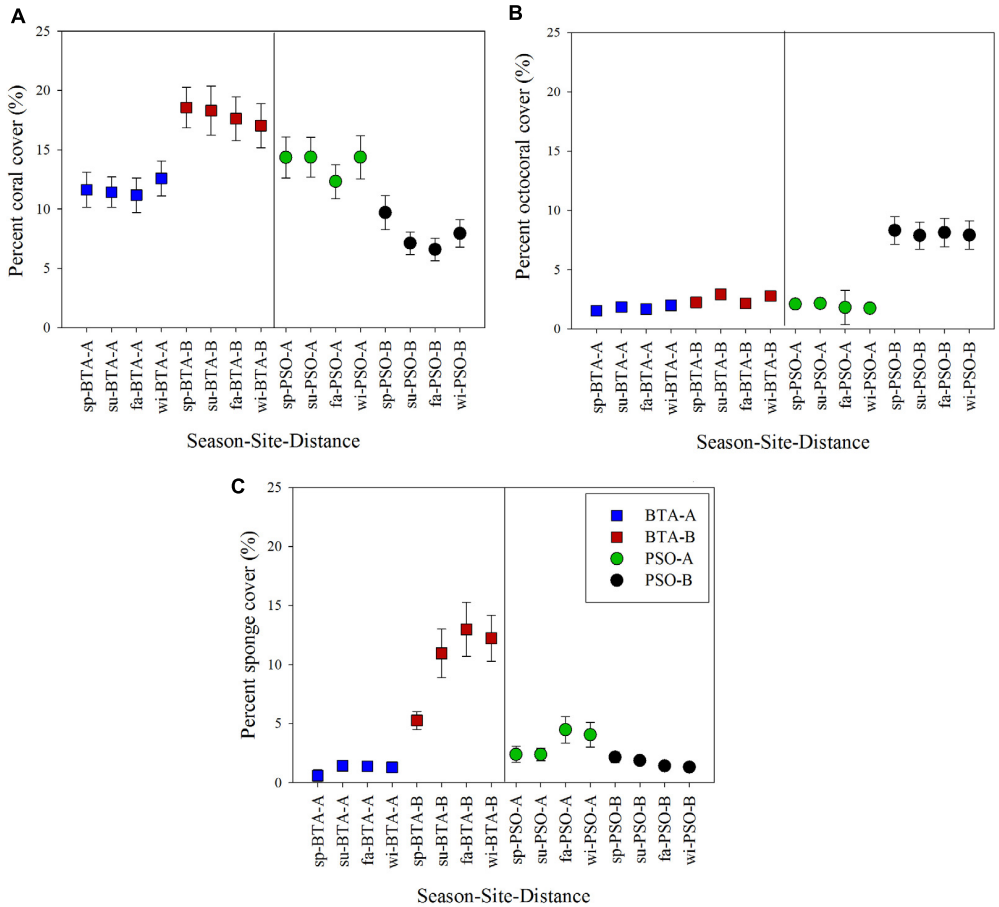


FIGURE 7 | Coral reef benthic components cover across a distance gradient from shore and seasons (mean ± 95% CI). Percent coral cover (Scleractininas + hydrocorals) **(A)**, octocoral cover **(B)**, and sponge cover **(C)**. Blue and red squares represent Bahía Tamarindo (BTA); green and black represent Punta Soldado (PSO). Seasons are defined as: sp, spring; su, summer; fa, fall; wi, winter.

while silt-clay had a negative significant association with coral colony abundance at PSO reef (**Appendix 1**). These patterns suggest that sediment texture differed significantly between sites (PERMANOVA, Pseudo F = 31.26, p = 0.001) and for the interaction site by distance from shore (Pseudo F = 5.52, p = 0.030). Multiple linear regressions analyses (visualized in dbDRA diagram) identified sand sediment particle distribution (r^2 = 0.33, p = 0.0007) as the most significant environmental variable that explained 49% of the total spatial and temporal variation of coral abundance at nearshore reefs (**Figure 11A**).

Coral recruit abundance had a strongly significant negative relationship with sand (RELATE, Rho = 0.225, p = 0.017) silt-clay (Rho = 0.24, p = 0.021), and depth (Rho = 0.715, p = 0.032) (**Table 3**). Multiple linear regression analysis showed that silt-clay (r^2 = 0.18, p = 0.020) had a significant association with coral recruit spatial and temporal array. Furthermore, coral recruit abundance also had a strongly significant relationship with wave height (r^2 = 0.52, p = 0.0270) and depth (r^2 = 0.45, p = 0.0250) for season by site interaction (**Figure 10B**). Vectors superimposed showed that coral recruit abundance was mostly influenced by silt-clay (<63 μm) and depth at PSO, where higher recruit abundance was documented at shallower areas closer to shore (**Figure 11B**). On the other hand, wave height had a stronger correlation to recruit abundance at BTA during the winter season, combined to a lesser extent with total precipitation. These environmental associations explain 87% of the total variation for coral recruit abundance. During the study period, peak SST reached up to 30°C, representing an anomaly of +1.35°C in relation to the mean monthly maximum, and might have also contributed to negatively impact coral abundance through undetected coral tissue loss, as coral decline followed high SST episodes. No bleaching was observed during the study period. However, there was no significant relationship between coral recruit abundance and SST (r^2 = 0.71, p = 0.37).

The non-parametric correlation BEST BIOENV (Spearman rank) analyses identified two groups of sediment variables that best correlated with coral reef benthic components, composed of sand and carbonate (Rho = 0.42), and sand, organic matter, carbonate, and terrigenous sediment (Rho = 0.412). Changes in percentage coral cover through the study period, 2014 to 2015, correlated with the variation of sand, silt-clay, carbonate, and terrigenous sediment content (p < 0.05) (**Table 4**). Based on our

TABLE 3 | Summary of RELATE (spearman rank) correlation matrix for coral colony abundance, coral recruit abundance and environmental variables.

	Coral colony abundance	Coral recruit abundance
Season-site-distance		
Sedimentation rate	Rho = 0.031, p = 0.3280	Rho = −0.127, p = 0.7636
Terrigenous rate	Rho = 0.029, p = 0.3477	Rho = −0.070, p = 0.5928
Sand	Rho = 0.475, p = **0.0030**	Rho = 0.225, p = **0.0278**
Silt-clay	Rho = 0.422, p = **0.0013**	Rho = 0.240, p = **0.0167**
Organic matter	Rho = 0.041, p = 0.2951	Rho = 0.028, p = 0.3836
Carbonate	Rho = 0.263, p = **0.0263**	Rho = −0.002, p = 0.4329
Terrigenous	Rho = 0.328, p = **0.0072**	Rho = 0.056, p = 0.2947
Site-distance		
Sedimentation rate	Rho = −0.086, p = 0.4950	Rho = −0.143, p = 0.6567
Terrigenous rate	Rho = −0.086, p = 0.5018	Rho = −0.143, p = 0.6714
Sand	Rho = 0.143, p = 0.3750	Rho = −0.029, p = 0.5023
Silt-clay	Rho = 0.143, p = 0.3729	Rho = −0.029, p = 0.5001
Organic matter	Rho = 0.086, p = 0.4156	Rho = −0.429, p = 0.7887
Carbonate	Rho = 0.543, p = 0.0834	Rho = 0.371, p = 0.2497
Terrigenous	Rho = 0.086, p = 0.3767	Rho = −0.143, p = 0.6645
Season-site		
Total precipitation	Rho = −0.157, p = 0.7964	Rho = −0.114, p = 0.7149
Wind speed	Rho = −0.253, p = 0.9546	Rho = −0.249, p = 0.8867
Wave height	Rho = −0.278, p = 0.9659	Rho = 0.266, p = 0.1065
SST	Rho = −0.180, p = 0.8541	Rho = −0.192, p = 0.8831
SST max	Rho = −0.232, p = 0.9459	Rho = 0.026, p = 0.4539
SST min	Rho = −0.241, p = 0.9573	Rho = 0.181, p = 0.2380
Depth	Rho = 0.858, p = **0.0322**	Rho = 0.715, p = **0.0275**

Averaged by season, site, distance, site by distance, and season by site. Bold p-values represent significance.

temporal and spatial analysis, there was a significant negative relationship between coral and macroalgal cover (r^2 = 0.70, $p < 0.0001$). A similar pattern was documented between coral and octocoral cover (r^2 = 0.462, p = 0.003). Spatial and temporal variations of macroalgal cover had a significant relationship with most of the sediment variables ($p < 0.05$), except organic matter (**Table 4**). Depth gradient across the reefs was correlated with macroalgal cover between seasons by site (Rho = 0.71, p = 0.036).

Sponge cover variation at nearshore reefs had a strong correlation with most of the sediment characteristics assessed, including sedimentation rate (Rho = 0.433, p = 0.002) and terrigenous rate (Rho = 0.371, p = 0.0001) (**Table 4**). Crustose coralline algae had similar correlations and had significant relationships to sedimentation rate (Rho = 0.256, p = 0.042), terrigenous rate (Rho = 0.295, p = 0.042), among other sediment dynamics, and depth ($p < 0.05$) (**Table 3**). Increased cyanobacteria cover, especially at BTA farther from shore, correlated with increased organic matter content (Rho = 0.262, p = 0.012) for the interaction site by distance from shore.

DISCUSSION

Near-shore coral reefs are experiencing rapid and significant spatial and temporal ecological changes in apparent response to human-induced degradation of coastal watersheds and interconnected coastal and marine habitats. In the context of current and forecasted climate change trends, such influences become increasingly significant. This study was primarily correlational and does not prove cause and effect. However, in the absence of other major disturbances during the study period of time (i.e., bleaching, disease outbreaks, hurricanes), documented land-based sediment pulses may have been one of the most important factors causing damaging effects on corals. Documented spatio-temporal patterns of coral reef benthic community structure near shore (<120 m from shore) had associations with sedimentation dynamics and interacting hydrodynamic forces. Sediment characteristics had an important role shaping benthic communities across a distance gradient from shore were sediment texture, categorized as silt-clay (<63 μm) and sand (>63 μm), and terrigenous sediments content (**Tables 3**, **4**). Results from this study suggest that significant spatio-temporal differences at near-shore coral reefs were associated with changing environmental conditions, in combination with increasing influences of coastal hydrodynamics.

Coral colony abundance in this study showed significant spatial and temporal differences and it was significantly associated with sand distribution patterns. Abundance decline during fall and winter seasons was related to the occurrence of extreme and acute weather events caused by a tropical trough and tropical storms that impacted the northern Caribbean region from August to November 2014. These atmospheric events produced the combined effect of sediment-laden runoff from disturbed coastal watersheds and increased wave action on shallow reefs (Otaño-Cruz et al., 2017). Coral reef benthic community structure and its ecological response is not only associated with variation in land-derived sediment input and distribution dynamics but is also related to bathymetry, as well as changes in local weather and oceanographic conditions. Weather and oceanographic dynamics that influence wave height and energy are the main drivers of sand sediment resuspension and transport (Hernández-Cruz et al., 2009; Field et al., 2011; Edmunds and Gray, 2014).

Coral species richness and diversity declined in a short-term period, especially at areas subjected to recurrent runoff and sedimentation stress. Further, seasonal variation was observed in multiple parameters, which also fluctuated between sites and distance zones, suggesting combined effects of natural seasonal variability at the studied spatial scales and terrestrial influences associated with changing land use and human-influenced runoff pulse dynamics. The scleractinian species recorded in BTA and PSO shallow reefs approximately represent two-thirds of the total number of species known for the northeastern region of Puerto Rico (Hernández-Delgado, 2000). The most conspicuous species at reef areas under continual sediment stress were *P. astreoides* and *P. porites*. The *Porites* spp. complex and the *Siderastrea* spp. complex have been recognized as species with high tolerance to sediment due to their ability to effectively reject particles, and they are becoming more abundant on shallow reefs throughout the Caribbean (Loya, 1976; Cortés and Risk, 1985; Torres and Morelock, 2002; Green et al., 2008; Ennis et al., 2016). Other common coral species at both study sites were *A. cervicornis,*

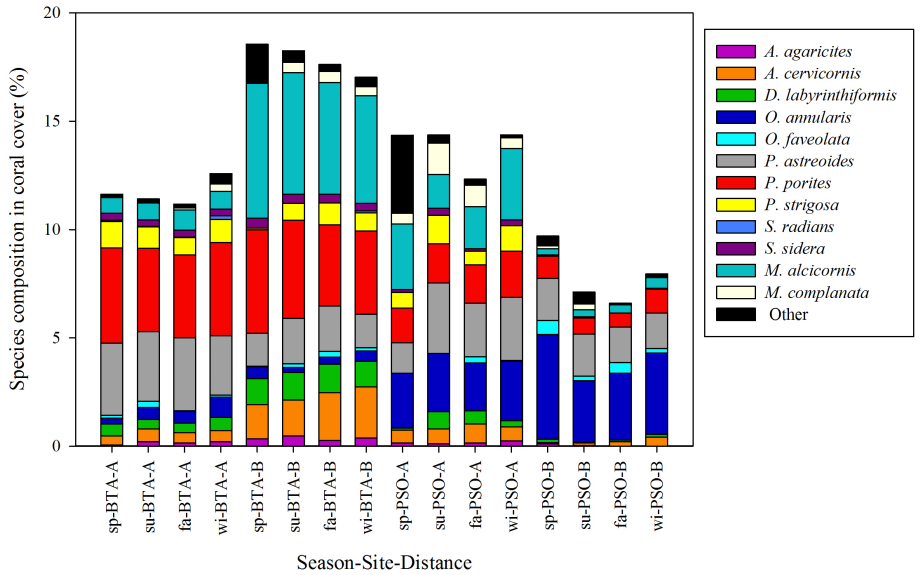


FIGURE 8 | Percent species composition of Scleractinian and hydrocoral species. Most dominant species among site, distance zone and seasons identified by colors. Seasons are defined as: sp, spring; su, summer; fa, fall; wi, winter.

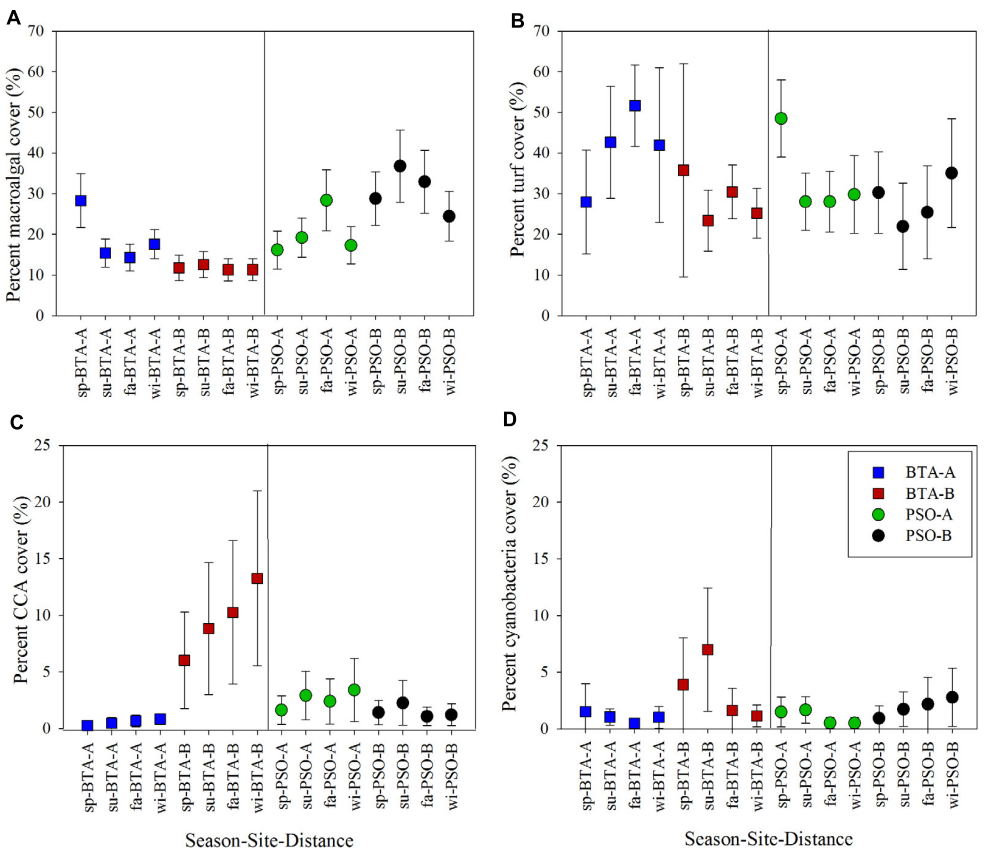


FIGURE 9 | Coral reef benthic components cover across a distance gradient from shore, site, and seasons (mean ± 95% CI). Percent macroalgal cover **(A)**, turf **(B)**, crustose coralline algae (CCA) **(C)**, and cyanobacteria cover **(D)**. Blue and red squares represent Bahía Tamarindo (BTA); green and black represent Punta Soldado (PSO). Seasons are defined as: sp, spring; su, summer; fa, fall; wi, winter.

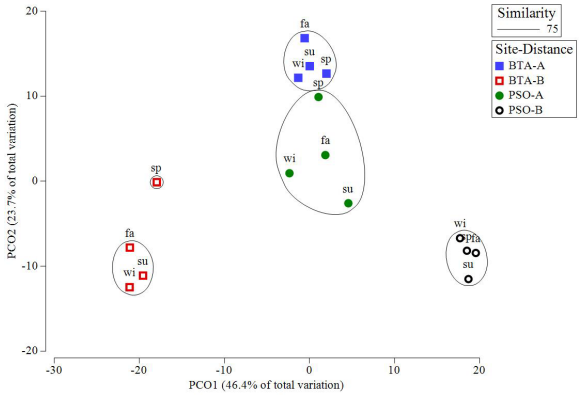


FIGURE 10 | Principal coordinates ordination (PCO) plot of benthic community structure on site by distance based on Bray-Curtis similarity matrix. Cluster represent 75% of similarity within groups. This model explained 75% of the observed spatio-temporal variation in benthic cover. Blue and red squares represent Bahía Tamarindo (BTA); green and black represent Punta Soldado (PSO). Seasons are defined as: sp, spring; su, summer; fa, fall; wi, winter.

D. labyrinthiformis, and *O. annularis,* abundant at reef zones less subjected to sedimentation stress (**Figure 3**). Transplanting, that was conducted prior to the initiation of this research, produced a high abundance of *Acropora* spp. across the study sites, showing that these species successfully established in areas less exposed to waves and land-based anthropogenic stress (**Figures 3**, **8**).

Rare and sensitive coral species have disappeared from locations affected by chronic sedimentation regimes across northeastern Puerto Rico (Hernández-Delgado, 2000). Over the last few decades, multiple Caribbean coral reefs have shown similar trends of declining coral diversity and coral abundance at sites impacted by recurrent pulses of sedimentation and LBSP (Loya, 1976; Cortés and Risk, 1985; Pastorok and Bilyard, 1985; Acevedo et al., 1989; Ennis et al., 2016). LBSP effects on coral reefs do not always reflect a discrete spatial gradient from shore since it mostly depends on the characteristics of marine sediment deposited across the reef (Pastorok and Bilyard, 1985) and on the ecosystem capability to process and adapt to sediment, organic matter and nutrient influxes (Lirman and Fong, 2007; Olds et al., 2018). Such changes might have adverse long-term repercussions due to reduced reef accretion and increased vulnerability to future and extreme climate scenarios (Knowlton, 2001; Edmunds, 2010; Hoegh-Guldberg et al., 2014). Degradation of the reef's structural complexity could further affect social-ecological services provided to coastal communities, including sustaining fish assemblages (Alvarez-Filip et al., 2009; Graham and Nash, 2013; Newman et al., 2015).

Coral recruit abundance of stress-tolerant species, such as *S. radians*, *S. siderea*, and *P. astreoides,* have become dominant in this study, as in many other locations in Puerto Rico. The increasing dominance of brooder species recruits has also been documented across the wider Caribbean (Lirman and Fong, 2007; Green et al., 2008; Edmunds, 2010; Hernández-Delgado et al., 2014a), which could be, not only the result of local environmental changes, but also the result of large-scale phenomena, such as climate change. This trend might be responsible for the overall shift in species composition as community trajectory is skewed toward ephemeral, fast growing, and stress-tolerant species. Under such regime, coral reefs experience a multiplicity of threats and recurrent changes in environmental conditions, thus limiting the success of reef-building species recruit settlement, such as *O. annularis* and *Acropora* spp. (Hernández-Delgado, 2000; Van Woesik and Jordán-Garza, 2011). In PSO reef, coral recruit abundance declined on the reef zone farthest from shore (>60 m) and it had a significant relationship with increased the proportion of silt-clay sediment deposition after an extensive deforestation event that disturbed the adjacent coastal watershed. Increased sediment influx and distribution of fine sediments through PSO reef was documented after strong precipitation events that were followed by wind-induced waves and currents that transported fine, land-based sediments until they were deposited in calmer waters (Otaño-Cruz et al., 2017). The effects of cold front events, characteristic of the winter season across the northern Caribbean, can produce strong long-period swells and a significant increase in sedimentation rate by both sediment bedload transport and resuspension (Otaño-Cruz et al., 2017). Previous studies have shown similar patterns of reduced or inhibited recruitment under sediment stress (Pastorok and Bilyard, 1985; Edmunds and Gray, 2014).

Other factors that affect coral recruit abundance are thermal stress (Edmunds, 2004; Van Woesik and Jordán-Garza, 2011), out-competition by fast-growing macroalgae (Nugues et al., 2004), and cyanobacteria (Fong and Paul, 2014). Algal dominance can inhibit coral larval settlement, predominantly when combined with deposited sediments, disturbing long-term coral reef resilience and their ability to recover after disturbance (Birrell et al., 2005; Kuffner et al., 2006; Vermij, 2006; Fong and Paul, 2014; Stubler et al., 2016). Extreme precipitation events have also been identified as a key factor that can trigger land-derived sediment pulses and distribution along near-shore coral reefs (Otaño-Cruz et al., 2017). Increased frequency of extreme rainfall events can have deleterious impacts on local coral populations and on nearshore coral rehabilitation efforts (Hernández-Delgado et al., 2014b). This combination of factors operating at multiple temporal and spatial scales can interact as long-term drivers of coral reef species composition change and should be further addressed.

Trends of live coral cover decline worldwide have often been associated with synergistic and complex local and regional chronic factors (i.e., disease, sedimentation, LBSP, increasing SST). Their combined effects with natural stochastic factors (i.e., hurricanes) have resulted in the long-term loss of primary reef-building species (i.e., *O. annularis* complex) and in shifts in species composition through the last few decades (Pandolfi et al., 2003; Rogers and Miller, 2006; Knowlton and Jackson, 2008; Miller et al., 2009; Hernández-Pacheco et al., 2011). Although the primary cause of coral decline in the Caribbean has been coral disease (Aronson and Precht, 2001a,b; Weil and Rogers, 2011), terrestrial sediment input is one of the main threats to near-shore coral reefs (Ogston et al., 2004; Hernández-Cruz et al., 2009; Ennis et al., 2016; Otaño-Cruz et al., 2017). Documented

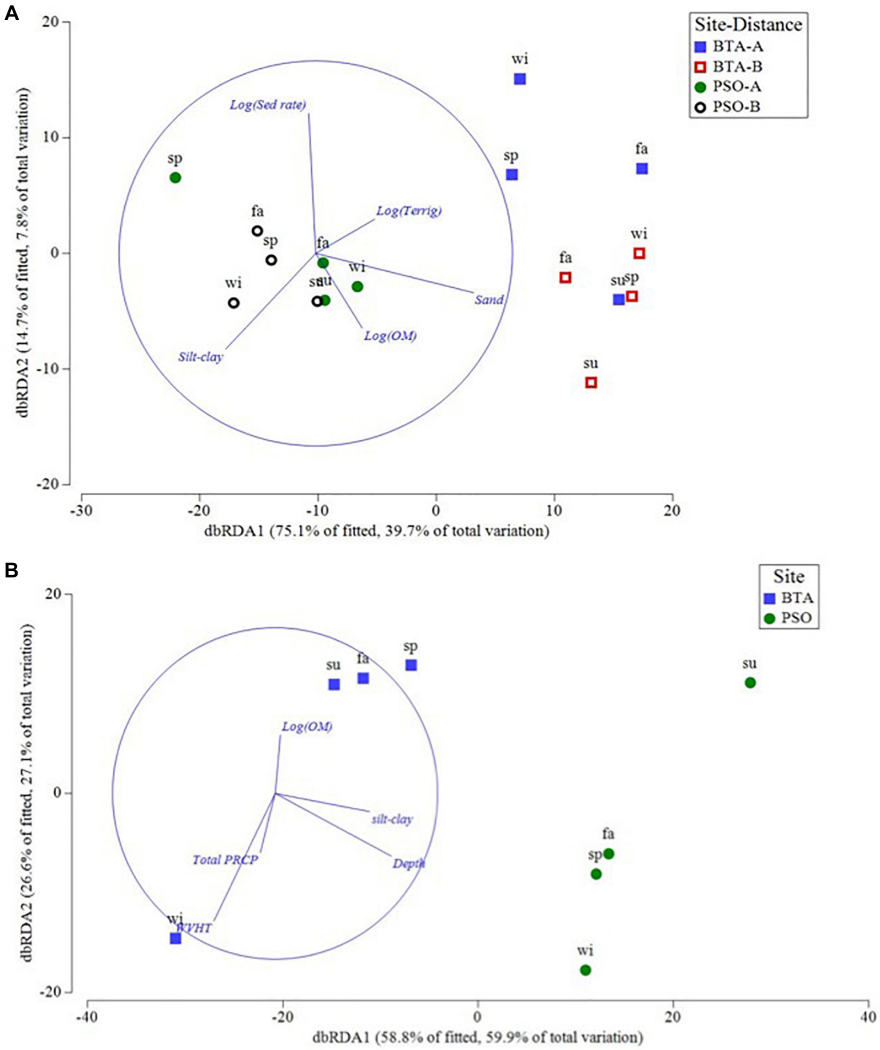


FIGURE 11 | Plot of redundancy analysis (dbRDA) for coral colony abundance **(A)** and coral recruit abundance **(B)**. Vectors represent environmental variables that best explains variation in coral abundance structure **(A)** between site by distance zone, and in coral recruit abundance **(B)** between sites. Blue and red squares represent Bahía Tamarindo (BTA); green and black represent Punta Soldado (PSO). Seasons are defined as: sp, spring; su, summer; fa, fall; wi, winter.

differences in percentage coral cover near shore were mostly associated with local sedimentation dynamics, principally with sand and silt-clay sediment distribution patterns. Coral cover decline at reef areas subjected to higher terrigenous sediment accumulation suggests an important relationship between coral reef health and changes in coastal watershed management, weather and local oceanographic dynamics. Documented mean live coral cover of 12.82% positions Culebra Island nearshore reefs below mean coral cover reported for the Caribbean region of 16.80% (Jackson et al., 2014), though this study was limited to shallow fringing reef systems which are often characterized by lower mean percentage live coral cover. However, the observed inverse relationship between live coral and macroalgal cover at nearshore reefs is a strong ecological indicator of degraded water quality conditions resulting from runoff pulse events and other land-derived stressors. Results from this study validate previous studies conducted across the Caribbean region that have recognized that increased human activities and unsustainable development along coastal watersheds can have major negative consequences on live coral cover and coral reef benthic community structure (Acevedo et al., 1989; Fabricius, 2005; Smith et al., 2008; Risk and Edinger, 2011; Bégin et al., 2013; Oleson et al., 2018).

Sediment accumulation on coral surfaces, especially fine sediments, can produce significant adverse physiological responses as a consequence of energy relocation, required to achieve rejection of sediment particles through the production of mucus and ciliary action (Acevedo et al., 1989; Telesnicki and Goldberg, 1995; Woolfe and Larcombe, 1999; Fabricius, 2011). Coral abilities differ between species and coral morphologies, with branching, meandering, and large coral colonies being more tolerant to sediment accumulation (Rogers, 1990;

TABLE 4 | Relate (Spearman rank) correlation of benthic components and environmental variables.

Variables	Coral cover	Octocoral	Sponge	Macroalgae	Turf	CCA	Cyanobacteria
Season-site-distance							
Sedimentation rate	Rho = −0.063	Rho = −0.093	Rho = 0.433	Rho = 0.001	Rho = 0.077	Rho = 0.256	Rho = 0.156
	p = 0.6698	p = 0.8067	p = **0.0011**	p = 0.4821	p = 0.2319	p = **0.0406**	p = 0.1038
Terrigenous rate	Rho = −0.073	Rho = −0.060	Rho = 0.371	Rho = 0.021	Rho = 0.026	Rho = 0.295	Rho = 0.173
	p = 0.6801	p = 0.6763	p = **0.0015**	p = 0.4340	p = 0.3702	p = 0.0380	p = 0.0862
Sand	Rho = 0.380	Rho = 0.290	Rho = 0.469	Rho = 0.308	Rho = 0.086	Rho = 0.285	Rho = 0.038
	p = **0.0014**	p = **0.0069**	p = **0.0003**	p = 0.0044	p = 0.1657	p = 0.0104	p = 0.2999
Silt-clay	Rho = 0.293	Rho = −0.264	Rho = 0.421	Rho = 0.260	Rho = 0.082	Rho = 0.241	Rho = 0.002
	p = 0.0075	p = 0.0116	p = 0.0003	p = **0.0122**	p = 0.1771	p = **0.0194**	p = 0.4416
Organic matter	Rho = 0.096	Rho = −0.147	Rho = −0.013	Rho = −0.07	Rho = −0.012	Rho = 0.16	Rho = 0.262
	p = 0.1693	p = 0.9597	p = 0.5111	p = 0.7495	p = 0.4974	p = 0.0795	p = **0.0151**
Carbonate	Rho = 0.255	Rho = 0.272	Rho = 0.467	Rho = 0.425	Rho = 0.065	Rho = 0.303	Rho = 0.065
	p = **0.0325**	p = **0.0150**	p = **0.0004**	p = **0.0003**	p = 0.2446	p = **0.0237**	p = 0.2599
Terrigenous	Rho = 0.261	Rho = 0.165	Rho = 0.402	Rho = 0.208	Rho = 0.061	Rho = 0.288	Rho = 0.080
	p = **0.0151**	p = 0.0559	p = **0.0008**	p = **0.0323**	p = 0.2348	p = **0.0114**	p = 0.1887
Site-distance							
Sedimentation rate	Rho = −0.086	Rho = −0.143	Rho = 0.543	Rho = 0.143	Rho = 0.771	Rho = 0.714	Rho = −0.200
	p = 0.4956	p = 0.6670	p = 0.2108	p = 0.4918	p = 0.2050	p = 0.1671	p = 0.6637
Terrigenous rate	Rho = −0.086	Rho = −0.143	Rho = 0.543	Rho = 0.143	Rho = 0.771	Rho = 0.714	Rho = −0.200
	p = 0.5044	p = 0.6666	p = 0.2094	p = 0.4882	p = 0.2096	p = 0.1637	p = 0.6660
Sand	Rho = 0.143	Rho = −0.029	Rho = 0.257	Rho = 0.257	Rho = 0.314	Rho = 0.371	Rho = −0.371
	p = 0.3667	p = 0.5071	p = 0.2917	p = 0.3367	p = 0.2032	p = 0.2526	p = 0.7970
Silt-clay	Rho = 0.143	Rho = −0.029	Rho = 0.257	Rho = 0.257	Rho = 0.314	Rho = 0.371	Rho = −0.371
	p = 0.3846	p = 0.4981	p = 0.2950	p = 0.3311	p = 0.2056	p = 0.2464	p = 0.7888
Organic matter	Rho = 0.086	Rho = −0.429	Rho = 0.257	Rho = 0.543	Rho = −0.086	Rho = 0.200	Rho = 0.829
	p = 0.4142	p = 0.7911	p = 0.4460	p = 0.2073	p = 0.4546	p = 0.3299	p = **0.0455**
Carbonate	Rho = 0.543	Rho = 0.371	Rho = −0.086	Rho = 0.429	Rho = 0.029	Rho = −0.029	Rho = −0.486
	p = 0.0855	p = 0.2452	p = 0.7077	p = 0.1297	p = 0.4965	p = 0.4926	p = 0.9178
Terrigenous	Rho = 0.086	Rho = −0.143	Rho = 0.486	Rho = 0.314	Rho = 0.486	Rho = 0.600	Rho = −0.314
	p = 0.3824	p = 0.6647	p = 0.1636	p = 0.3280	p = 0.2509	p = 0.1698	p = 0.7896
Season-site							
Total precipitation	Rho = −0.062	Rho = 0.048	Rho = 0.313	Rho = −0.005	Rho = −0.259	Rho = −0.083	Rho = 0.032
	p = 0.5636	p = 0.3195	p = 0.0762	p = 0.4367	p = 0.9659	p = 0.6343	p = 0.3503
Wind speed	Rho = −0.197	Rho = −0.194	Rho = −0.152	Rho = −0.067	Rho = 0.126	Rho = −0.178	Rho = 0.039
	p = 0.8690	p = 0.8599	p = 0.7491	p = 0.5824	p = 0.1950	p = 0.8221	p = 0.3203
Wave height	Rho = −0.270	Rho = −0.124	Rho = −0.073	Rho = −0.276	Rho = −0.131	Rho = −0.035	Rho = −0.048
	p = 0.9477	p = 0.7138	p = 0.6061	p = 0.9173	p = 0.6478	p = 0.5341	p = 0.5128
SST	Rho = −0.151	Rho = −0.189	Rho = 0.044	Rho = −0.248	Rho = 0.093	Rho = −0.212	Rho = −0.08
	p = 0.7957	p = 0.8776	p = 0.3595	p = 0.9616	p = 0.2764	p = 0.9322	p = 0.5629
Depth	Rho = 0.858	Rho = 0.759	Rho = 0.456	Rho = 0.706	Rho = −0.009	Rho = 0.759	Rho = −0.17
	p = **0.0284**	p = **0.0272**	p = 0.0625	p = **0.0274**	p = 0.5703	p = **0.0319**	p = 0.8587

Bold p-values represent significance.

Fabricius, 2005; Sanders and Baron-Szabo, 2005). Field studies conducted at Costa Rica (Cortés and Risk, 1985) and southwest Puerto Rico (Torres and Morelock, 2002) documented that the *O. annularis* complex experienced significant reduction of growth rates and live tissue cover with increased terrigenous sediment accumulation. Similar trends were evidenced in this study where massive *O. annularis* composed the lowest live coral cover, and in contrast, *P. astreoides* and *P. porites* had the highest benthic cover. Benthic cover dominance by species that can survive in sub-optimal conditions suggests that these reefs have already experienced changes due to a chronic, recurrent land-based stressors.

Other benthic components that correlated with terrigenous sediment deposition were macroalgae, calcareous algae, and sponge percentage cover. Macroalgae, turf, and calcareous algae cover experienced variation in benthic cover in a short time period, particularly the fast-growing *Dictyota* spp. and encrusting calcareous algae *Ramicrusta textilis*, which were documented overgrowing dead coral skeleton. In 2011, *R. textilis* was first documented in Puerto Rican coral reefs (Ballantine et al., 2011;

Ballantine and Ruiz, 2013), and continues to outcompete and overgrow multiple coral species (Ballantine et al., 2016). This encrusting species has been documented overgrowing at least 14 species of scleractinian corals, gorgonians, hydrocorals and other algae (Eckrich et al., 2010) and has become a critical factor adversely influencing coral assemblages across the Caribbean region. Its role should be carefully studied. Increased sediment deposition could inhibit fish grazing and promote the growth of macroalgae and filamentous algal turfs (Bellwood and Fulton, 2008). Macroalgae can also proliferate under high nutrient concentration from runoff (Cloern, 2001), under low herbivory pressure due to the slow recovery of *Diadema antillarum* (Ruiz-Ramos et al., 2011; Rodríguez-Barreras et al., 2018), overfishing (Hernández-Delgado et al., 2006), or due to a combination of any of these factors (Littler et al., 2006). This suggests that coral reef trophic condition is also a critical co-factor, in combination with climate change-related impacts, in shaping coral reef benthic assemblages.

Increased frequency and severity of sedimentation stress and LBSP can increase sponge competition for space and could favor benthic cover of resistant species, thus leading to alternate dominant states (López-Victoria et al., 2006; Chadwick and Morrow, 2011; González-Rivero et al., 2011). Even though the increased cover of encrusting sponge *D. funicularis* has been recently reported on Caribbean reefs (García-Sais et al., 2016; Kramer et al., 2016), its distribution and coral out-competition effects are still unknown. High octocoral cover documented in deeper areas at PSO concurs with previous studies that have shown increased dominance of octocorals at deeper reefs (Sánchez et al., 1997). Results from this study show that there was an inverse relationship between coral and octocoral cover.

A possible limitation of the study was the lack of resources to assess benthic community and environmental variables at a larger spatial scale to contrast various land uses and coastal management strategies, and the response of coral reefs. Future research could address larger spatial scales and multiple environmental factors affecting water quality, such as nutrient concentration, to identify which factors exert greater influence in coral abundance, coral recruit patterns, coral cover, and macroalgae cover at local and regional levels. It is also imperative to address potential effects of increased frequency of extreme events, including intense storms and wave action, in the context of changing land use and other localized human-driven influences. Hurricanes can have highly destructive mechanical impacts to coral reefs (Woodley et al., 1981; Fenner, 1991; Toledo-Hernández et al., 2018), and in the context of changing land use, they can magnify sediment delivery and resuspension along coastal coral reefs and associated ecosystems.

The potential influence of sedimentation in determining coral reef trajectories highlights the need for a broader understanding of sedimentation dynamics and of coral reef social-ecological responses under variable environmental scenarios, in order to inform decisions and policies to reduce local stressors, improve water quality, and effectively conserve and restore threatened coral reefs.

AUTHOR CONTRIBUTIONS

AO-C, AM-A, EH-D, EC, and JO-Z conceived and designed the study. AO-C, AM-A, NG-R, DD-M, and EB performed the coral reef monitoring, field sampling, and analysis. AO-C, AM-A, EH-D, EC, and JO-Z analyzed the data. AO-C, AM-A, and EH-D wrote the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00551/full#supplementary-material

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Traditional Ecological Knowledge Supports Ecosystem-Based Management in Disturbed Coastal Marine Social-Ecological Systems

Fernanda Terra Stori[1], Caiuá Mani Peres[1], Alexander Turra[1] and Robert L. Pressey[2]*

[1] Laboratory of Coastal and Marine Ecosystems Management, Oceanographic Institute, University of São Paulo, São Paulo, Brazil, [2] Australian Research Council Centre of Excellence for Coral Reef Studies, James Cook University, Townsville, QLD, Australia

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***Correspondence:**
Fernanda Terra Stori
f.terra.stori@gmail.com

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A growing number of studies suggest a participatory ecosystem approach to support decision-making toward resilience and sustainability in social-ecological systems. Social-ecological resilience (SER) principles and practices are recommended to manage natural crises. However, it is necessary to broaden our understanding of SER on human-induced disturbances driven by economic development projects. In this paper we present the social-ecological system of Araçá Bay (Brazil), a small-scale fishery community that has experienced successive disturbances due to development projects since the 1930s. There was a lack of studies about the impacts of development projects in this bay. As part of a major project that aimed to build an ecosystem-based management plan for Araçá Bay through a participatory planning process, we focused on investigating fishers' traditional ecological knowledge (TEK) to understand Araçá Bay's small-scale fisheries social-ecological system. The objectives were to: (1) investigate fishers' TEK regarding management practices and linked social mechanisms, human-induced disturbances and their consequences for the social-ecological system, ecosystem goods and services, and future threats; and (2) provide information based on TEK to the participatory planning process and analyze its contribution to Araçá Bay's ecosystem-based management plan. Combined methods were used during 3 years of intense research-action (2014–2017): in-depth ethno-oceanographic interviews with expert fishers; monitoring Araçá Bay participatory meetings; and participant observation. Genuine local practices and social mechanisms from traditional culture were recorded, as well as TEK about 57 target fish species and methods to protect habitats and natural resources. Fishers also reported ecosystem disturbances and recovery processes. TEK was codified through SWOT analysis to assist the participatory planning process. Ecosystem services and threats based on TEK were brought to the participatory process, acknowledged by the participants, and incorporated into the management plan. TEK analysis proved to be an important methodology to

provide historical environmental data regarding the impacts of development projects and support planning in disturbed ecosystems. In order to support coastal marine ecosystem-based management strategies toward SER and sustainability, researchers and practitioners should consider traditional territories in planning, recognize local practices and social mechanisms, and consider TEK on ecosystem goods and services and on historical human-induced disturbances.

Keywords: social-ecological system, social-ecological resilience, traditional ecological knowledge, coastal marine planning, ecosystem based management, Araçá Bay

INTRODUCTION

Innovative approaches suggest that adaptive and participatory management practices of common pool resources, supported by ecosystem-based management, are crucial to favor resilience and promote sustainability in social-ecological systems (Feeny et al., 1990; Berkes et al., 2003; Douvere, 2008; Gibbs, 2009; Biggs et al., 2012; Jones et al., 2013; Simonsen et al., 2015). Social-ecological systems integrate people and nature with reciprocal and interdependent feedback. To better understand the links between ecosystems and social systems, it is necessary to understand the relationship between different forms of management, considering ecosystem goods and services, the values they generate and their resilience (Foley et al., 2005; De Groot et al., 2010; Sartori and Monteiro, 2010; Moberg and Simonsen, 2014).

Social-ecological systems are likely to be exposed to disturbances such as storms, droughts, pests, and resource collapse. However, some systems seem to be more resilient than others (i.e., have the capacity to undergo disturbance and maintain their functions and controls) (Gunderson and Holling, 2002). Social-ecological resilience (SER) is, therefore, the magnitude of disturbance that can be tolerated by a social-ecological system before it moves to a different region of state space controlled by a different set of processes (Carpenter et al., 2001).

To measure the tolerated disturbance in a social-ecological system can be an uncertain task, especially when suitable data are not available. In this context, many studies recommend practices and principles based on an ecosystem approach to manage natural disturbances and build SER (Folke et al., 2003; Biggs et al., 2012; Simonsen et al., 2015). Notwithstanding, it is still necessary to broaden the understanding of SER practices and principles to overcome human-induced disturbances, such as those introduced by economic development projects such as ports, industries, mining, and energy plants.

Traditional ecological knowledge (TEK) can play a key role in providing information about natural and human-induced disturbances, and in supporting coastal marine ecosystem-based management strategies (Berkes et al., 1995; Moura and Diegues, 2009; Stori et al., 2012). TEK refers to a cumulative body of knowledge, practices, institutions, and beliefs, evolving by adaptive processes and handed down through generations by cultural transmission (tradition). TEK encompass worldviews developed and sustained by traditional communities in interaction with their biophysical environments (Gadgil et al., 1993; Berkes et al., 1995; Berkes and Folke, 1998; Berkes, 1999, 2004; Toledo, 2002; Colding et al., 2003; Gómez-Baggethun et al., 2013). TEK lies behind the adaptive capacity of many rural and indigenous communities that have historically been able to conserve biodiversity while enhancing livelihoods and adapting to disturbances (Ruiz-Mallén and Corbera, 2013).

A diversity of local management practices and linked social mechanisms based on TEK are recommended to promote the management of common-pool resources and foster SER and sustainability in a defined social-ecological system (Berkes and Folke, 1998; Folke et al., 1998, 2003; Berkes et al., 2000; Gunderson and Holling, 2002). Specifically, Folke et al. (1998) codified thirteen management practices based on TEK and four types of social mechanisms associated with those practices, which can foster SER. Such practices include, for instance, monitoring temporal or total protection of species or habitats, multiple species management, resource rotation, and social mechanisms including cross-scale institutions, taboos and regulations, rituals or ceremonies, and social and religious sanctions (Folke et al., 1998; Berkes et al., 2000).

This study examines fishers' TEK and analyzes its role in supporting ecosystem-based management strategies in an area historically disturbed by economic development projects: Araçá Bay (São Paulo State, Brazil). The data obtained were used to inform a participatory planning process developed to elaborate a management plan for the area. Specifically, the objectives were to: (1) investigate fishers' TEK regarding management practices and linked social mechanisms, human-induced disturbances and their consequences for the social-ecological system, ecosystem goods and services, and future threats; and (2) provide information based on TEK to the participatory planning process and analyze its contribution to Araçá Bay's ecosystem-based management plan.

In the following section we present the local context followed by a brief explanation about the participatory planning process carried out in Araçá Bay, and the methods conducted to assess the TEK and its application in the Araçá Bay management plan. We then report the TEK assessment and its contributions to the participatory process. The results are discussed considering the importance of TEK for participatory ecosystem-based management strategies that favor SER and

sustainability in coastal marine areas threatened by economic development projects.

MATERIALS AND METHODS

Study Area

São Paulo State has 248,220 km^2 of land area, an estimated population of 45.1 million (Instituto Brasileiro de Geografia e Estatística [IBGE], 2017). The São Paulo State gross domestic product of USD 581 billion, represents 33% of the Brazilian total (SEADE, 2017). São Paulo is a strategic state to Brazilian growth, and the development of port activities has fundamental importance to achieve this. The State has two ports, the Santos Port (the largest port of South America) and the São Sebastião Port, located in the vicinities of Araçá Bay, São Sebastião municipality in the Northern Coast of the state (**Figure 1**).

The northern coast is characterized by a high diversity of natural resources and great potential for tourism development, but is threatened by uneven land division, intense real estate speculation, overfishing, and the development of nautical and port activities (Secretaria do Meio Ambiente do Estado de São Paulo/Coordenadoria de Planejamento Ambiental [SMA/CPLA], 2012). Furthermore, in 2007, the Brazilian government announced the discovery of a large field of oil and natural gas in the Santos Basin, which could raise the oil reserves to 87 billion barrels (Magalhães and Domingues, 2014; Aloise de Seabra et al., 2015; Petrobrás, 2018). This growing activity has increased the demand for port infrastructures in the coastal zone and is threatening fragile ecosystems.

Araçá Bay is a well-defined geographic area, which contains essential ecosystem goods and services. This bay has experienced severe disturbances to its ecological state due to the many phases of port expansion and the impact of oil production, affecting traditional culture (namely *caiçara*) and tourism activities (Amaral et al., 2010, 2015; Peres et al., 2016). Despite the impacts that the port construction brought to Araçá Bay, it remains a unique environment in the São Paulo State coastal zone. It is a tide-dominated shallow mudflat area near a channel with depths up to 40 m, and presents a mix of ecosystems such

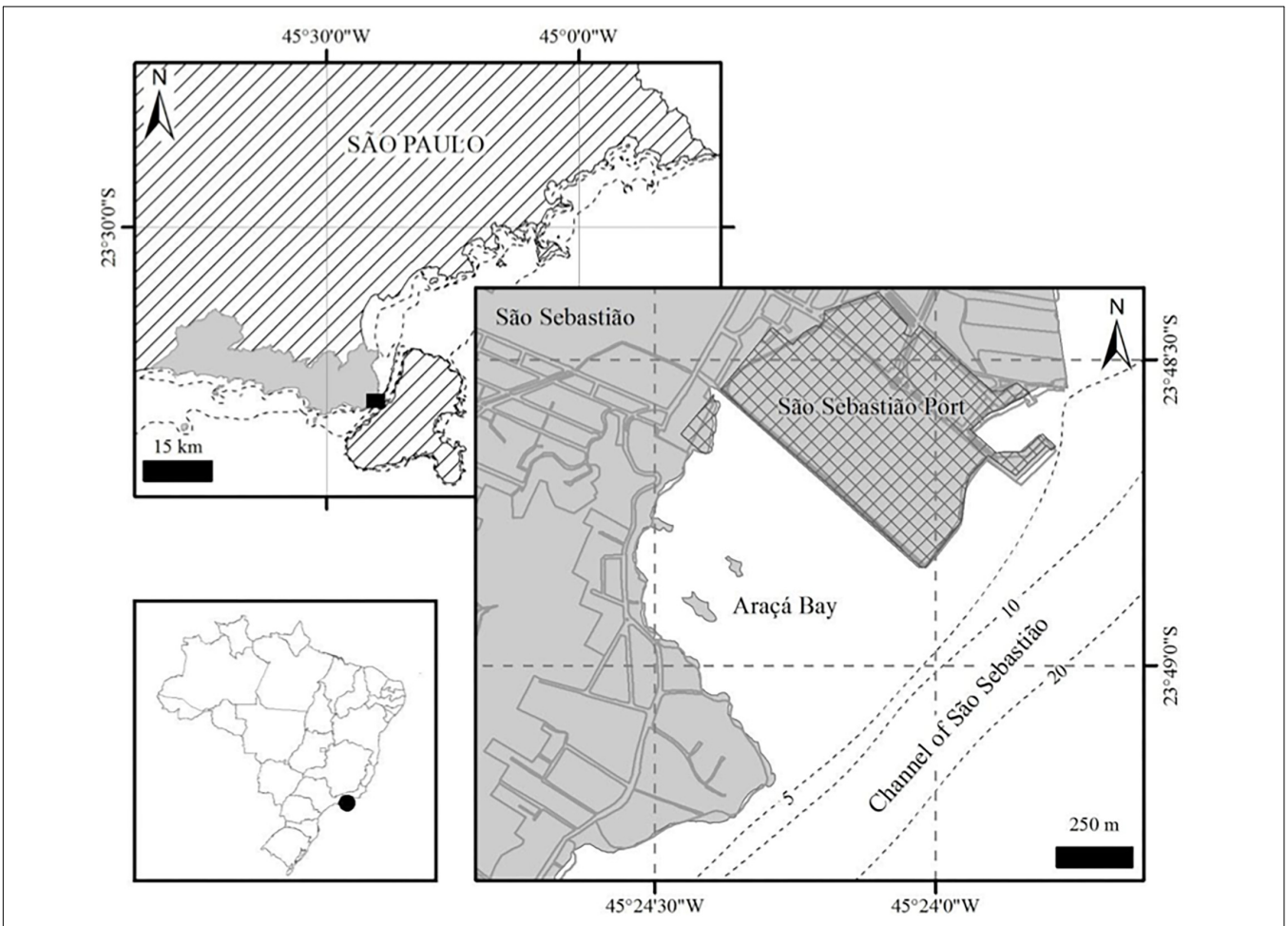


FIGURE 1 | Location of Araçá Bay on the northern coast of São Paulo State, Brazil. In the bottom left, the location of Araçá Bay in Brazil, represented by a black spot. In the **top left**, the location of São Sebastião municipality in the São Paulo State is represented in light gray, and Araçá Bay represented by a black square. Araçá Bay is represented in the **right**, and the area of the constructed port is represented by a grid (scheme kindly organized by Luciana Y. Xavier).

as mangroves, beaches, rocky shores, and islets. To date, more than 1,400 species have been described at Araçá Bay, which environments serves as an important breeding area for marine life and providing connectivity with other mangroves in the region (Amaral et al., 2015).

The São Paulo State Government created in 2008 three large marine protected areas (MPAs) along the entire coastal zone (Stori et al., 2019). At this point, Araçá Bay became encompassed by the Marine Protected Area of the Northern Coast of São Paulo (MPA-NC, State Decree N° 53.525, Governo, do Estado de São Paulo, 2008). However, the MPA-NC decree was not clear about the comprehensiveness of the Araçá Bay area within the MPA boundaries, leading to debates about whether economic developments projects could be placed there (**Figure 2**).

Araçá Bay is also governed by the Ecological-Economic Zoning of the Northern Coast (State Decree N° 62.913/2017, Governo do Estado de São Paulo, 2017), a binding instrument of the State Plan for Coastal Zone Management (State Law N° 10.019/1998, Governo do Estado de São Paulo, 1998). The zoning should consider abiotic and biotic structures, functions, as well as current and future uses and activities. Water standards, topography, protected areas, breeding areas, fishing activities, aquaculture, nautical activities and ports were the main attributes considered to classify the zones (Stori et al., 2019). This law adopted a multiple-use zone system that ranges from Z1M (the most preserved areas) to Z5M (areas impacted by urban-industrial activities).The maritime range inside Araçá Bay is classified as Z2M, while the classification varies in the intertidal range as follows: Z2M in the rocky shore; Z3M in the sandy shore and islets (in front of the urban area); and Z5M in the port area (**Figure 3**). The terrestrial range of the Ecological-Economic Zoning was classified as Z4 in the law in force from 2004 to 2017. However, in the process of zoning revision, the urban-industrial area was turned into a Z5 zone.

Araçá Bay comprises a marine area of 550,000 m^2 that was formed during the construction of São Sebastião Port (**Figures 1**, **3**). The port was built in four phases of land reclamation. The first phase, initiated in 1936 and finalized in 1955, was linked to the construction of an oil refinery by Petrobras, the major Brazilian oil company. The refinery construction was preceded by earthmoving works to gain more area, with voluminous excavations at the base of the mountain range, with the removed sediment used for port construction (Peres et al., 2016). The second phase was finished in 1973, the third phase in 1987, and the fourth phase was concluded in 1988 (Peres et al., 2016; Turra et al., 2017).

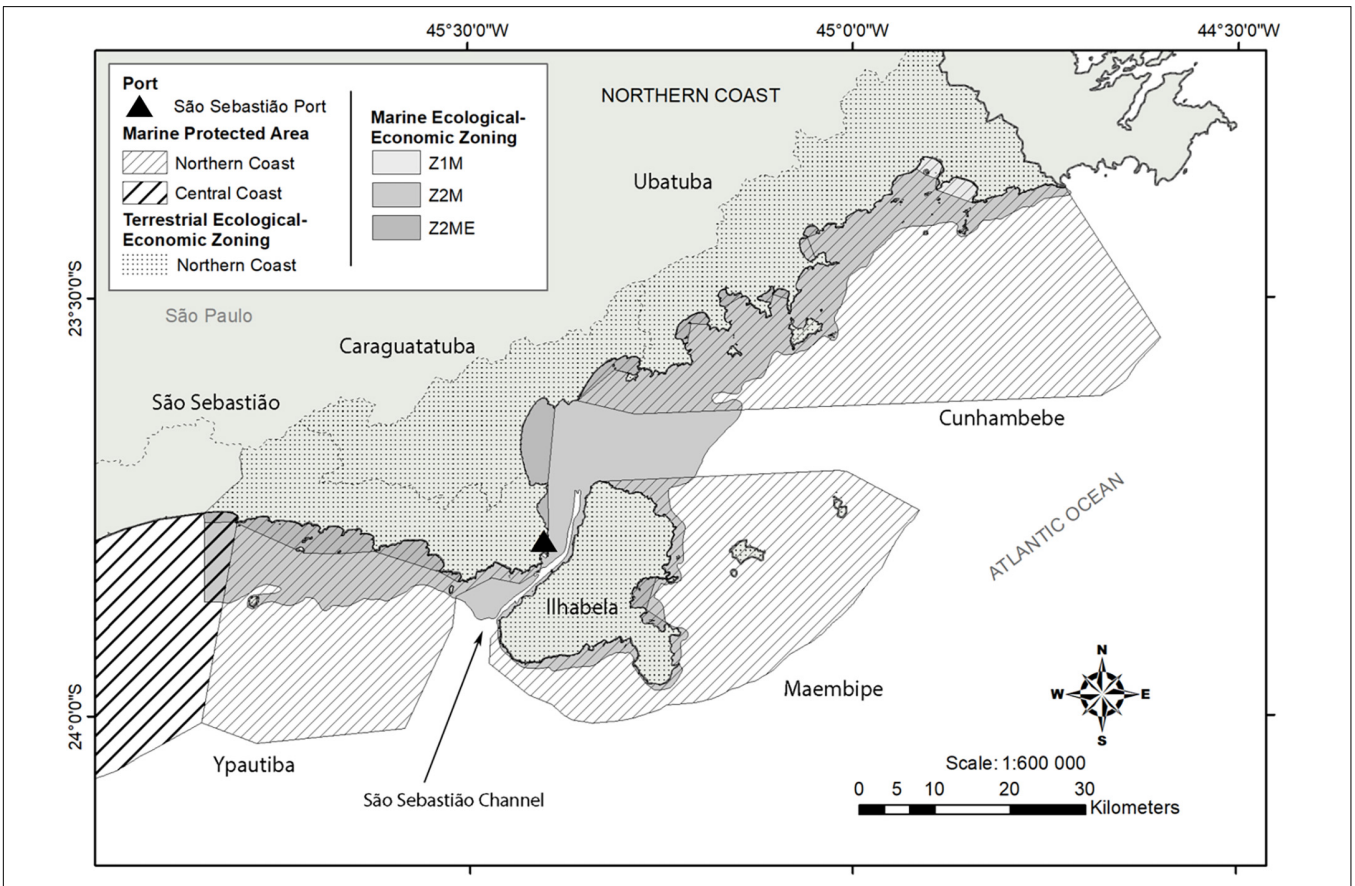


FIGURE 2 | The northern coast of the São Paulo State showing the Marine Protected Area of the Northern Coast (MPA-NC) (figure obtained from Stori et al., 2019). Araçá Bay is located along the São Sebastião channel, adjacent to the port (represented by the triangle symbol).

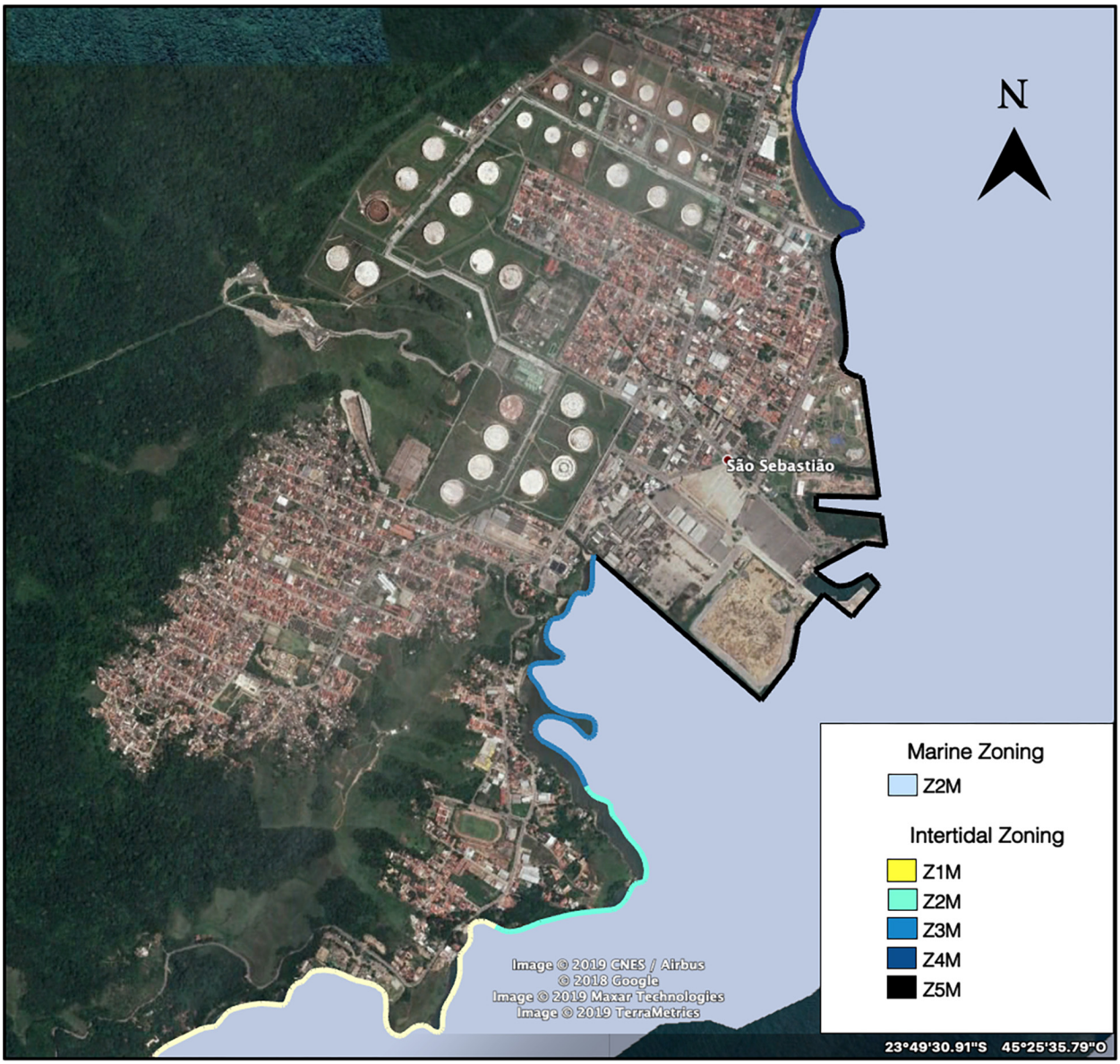


FIGURE 3 | The marine zoning of the State Plan for Coastal Zone Management in the central area of the Municipality of São Sebastião. In the marine range, Araçá Bay is classified as Z2M and, in the intertidal range, the coastline is classified in Z2M in the rocky shore, Z3M in the sandy shore and islets, and Z5M in the zone of the port. All the urban-industrial area, which includes residential occupation, the port and the Petrobrás (the round shapes are the oil storage tanks) are currently classified as Z5M (Image from Google Earth).

São Sebastião Port is a minor port in São Paulo State when compared to Santos Port, the largest port in Latin America, ranking at the 42nd global position (Lloyd's List, 2017). While Santos Port trades more than 3 million tons of products per year, São Sebastião Port trades nearly 700,000 tons (Porto de São Sebastião, 2019).

In order to increase the port's competitiveness, the São Sebastião Port Authority, a division of the Secretary of Logistics and Transports of the São Paulo State, applied in 2004 for an environmental license from the Brazilian Institute of Environment and Renewable Natural Resources (IBAMA), with the objective to expand the port land reclamation over 82% of Araçá Bay to serve as a container depot (Turra et al., 2017). This project would practically cover all the marine area left in the bay. Due to IBAMA recommendations and to public complains, the initial project (land reclamation) was replaced by the idea to build the port over a concrete slab supported by 17,000 piles, reducing the expansion area down to 34% (Turra et al., 2017). In December 2013, IBAMA approved the new project and granted the port authority a license to proceed with the port expansion. In 2016, the license was canceled due to a legal process jointly moved by Federal and State prosecutors, supported by the local community

and by the researchers involved in the participatory planning process of Araçá Bay. The judge stated that the authorities should wait for the results of the Biota-Fapesp/Araçá Project in order to further decide on the permanent elimination of part of the bay. According to the judicial decision, the port authority should start a new environmental impact assessment if the objective to expand the port persisted. More recently, the port authority announced the intention to fragment the expansion project into different stages and restart the licensing process (Porto de São Sebastião, 2019). However, the licensing process has not yet been initiated.

The Participatory Planning Process of Araçá Bay

The present research was part of the Integrated Management Group of the Biota-Fapesp/Araçá Project. Active from 2012 to 2018, the project aimed to understand in depth the functioning of Araçá Bay, with the collaboration of more than 170 researchers from 35 universities, distributed in 12 different research groups focused on subjects that comprised biology, chemistry, physics, and social sciences (Amaral et al., 2015).

In this context, the Integrated Management Group aimed to foster social learning by producing an ecosystem-based management plan for Araçá Bay, the so-called "Local Plan for Sustainable Development of Araçá Bay." The group adopted the scientific paradigm of Post-Normal Science to guide the planning (Stori et al., 2017a; Santos et al., 2018). This paradigm addresses complex situations, including uncertainties of facts, pluralities of opinions and values, and a pressing need for decision-making (Funtowicz and Ravetz, 1991, 1993, 1997). The Integrated Management Group's conceptual goal was to produce "*a science committed to the co-management of coastal marine common resources, that integrates formal and non-formal knowledge and institutions, stimulating the learning and strengthening of social networks to the social-ecological resilience for the sustainability of ecosystems goods and services*" (Stori et al., 2017a).

The core of the Integrated Management Group was composed of 13 researchers. However, the group also had the support of volunteers to organize the meetings. The researchers had common duties to plan, organize, lead the participatory activities, and evaluate the results of the participatory meetings (detailed information can be found in Turra et al., 2016). Additionally, each researcher had specific research to undertake about Araçá Bay, and to relate the results to the participatory process. The researchers produced studies on public policy analysis, the challenges of social participation, social learning, and the social-ecological system analysis presented in this manuscript.

In order to achieve the common objectives of the Integrated Management Group, the researchers developed diverse strategies to stimulate social participation. Participatory meetings to develop the Local Plan for Sustainable Development of Araçá Bay, and the establishment of a formal forum within the Marine Protected Area of the Northern Coast of São Paulo (MPA-NC) were the main strategies adopted. These approaches helped to create a relationship based on mutual respect and trust between community members and scientists.

Seven participatory meetings were organized from 2014 to 2016, aimed at developing the ecosystem-based management plan. A total of 141 people from different social sectors participated in the meetings, including representatives from the Araçá Bay community (fishers and non-fishers), citizens of other parts of the municipality or of the northern coast, researchers, students, representatives from non-governmental organizations, entrepreneurs, and representatives from the municipal government, MPA-NC, and port authority (Turra et al., 2016).

A detailed examination about the challenges of social participation in Araçá Bay is being produced by Santos et al. (unpublished results) and details about the mobilization strategy is available at Santos et al. (2018). In the first meeting, participants suggested the best venues and days of the week for the next events. Additionally, at the end of the meeting, participants were asked to give opinions about the positive and negative aspects of the participatory process. The high participation of scientists and the low participation of the Araçá Bay community, and of other São Sebastião residents and public authorities, was mentioned as a negative aspect of the first meeting. In order to promote wider social participation, the participants suggested intensifying the invitations to attract more participants. However, they were not able to carry out this action alone, so the research group was responsible for developing and performing the invitation strategies, while the participants were committed to inviting family members, friends, and neighbors (Santos et al., unpublished results).

The group of researchers tried different strategies to mobilize participants to the meetings, both at the scale of northern coast, and at the scale of the Araçá Bay neighborhood. The group compiled a mailing list with more than 1,000 e-mails targeting people and institutions interested in social and environmental issues on the northern coast, as well as official invitations to public authorities (Santos et al., 2018). Regarding the mobilization of people from the neighborhood, the researchers delivered the invitations "door to door," rented a sound car to announce the meetings, and installed a canopy tent in strategic locations to give information about the project (Santos et al., 2018).

The research group also took advantage of social media communication tools (e.g., Facebook and WhatsApp) to engage the community in the process. These tools assisted in organizing the meetings and helped to share information in the network, such as environmental news and complaints about environmental offenses (e.g., mangrove deforestation, irregular marine litter disposal, sewage contamination into the bay, shipping oil spills). Press releases were produced after each meeting and sent to the mailing list to inform people about the preliminary results and invite the participants to the next meetings (Santos et al., 2018). Despite all the efforts made to mobilize the Araçá community, not all those who were invited joined the participatory meetings (Santos et al., unpublished results). Also, the presence of people from other municipalities was low due to the long distances and difficulties in reaching the meetings on time. According to Santos et al. (unpublished results), participation is an act of will and the researchers were able to mobilize only the individuals who were motivated, i.e.,

those who exhibited interest, enthusiasm, and the determination to act toward a better quality of life.

Participants could suggest improvements to the participatory process at any time. For instance, they proposed a "kids' space" to facilitate adults' attendance, and suggested activities promoting *caiçara* culture, including canoe riding, organizing the meetings on the beach, and serving *caiçara* meals for the coffee break (Santos et al., 2018). The research group engaged a group of biology and oceanography students to create environmental games, so the children could play and learn while the adults were in the meetings (Santos et al., 2018). The researchers did not refuse any of the requests. However, they were not able to arrange all the meetings on the beach (only two meetings were organized on the beach) and to provide fresh fish in the coffee break as requested by some participants. Instead, the group found a wide and sheltered place for the meetings (a municipal public space) and provided fruits, juices, and traditional cakes for the coffee break.

Regarding the methods adopted, different participatory techniques were applied to obtain participants' opinions about the importance of Araçá Bay (the ecosystem goods and services), about the current problems and future threats, and to develop future scenarios and management actions (Turra et al., 2016; Santos et al., 2018). The researchers made a great effort to consider all the opinions during the participatory process and participants acknowledged learning, hope, union, strengthen, consensus, and integration between members as positive aspects of the meetings (Santos et al., unpublished results). At the seventh meeting, people were invited to form a group of stakeholders interested in the plan's implementation. The group was named by the participants as "The Guardians of Araçá Bay," currently with 25 members.

The scientists also helped the community to promote the traditional "*caiçara* canoe regatta." The regatta is a celebration promoted by the community with typical indigenous canoes, aiming to raise awareness about the importance of *caiçara* culture and its maintenance. The local community has promoted nine regattas so far, and the researchers helped them to organize three canoe regatta events (2014, 2015, and 2016).

Researchers also organized meetings to introduce the local community to the Federal and State prosecutors (the Public Ministry). During these meetings, participants discussed solutions to the main problems identified in the management plan (such as irregular litter and sewage disposal) and were informed about the pace of the legal appeal process for the cancelation of the port expansion. Leaders of the community organized a protest against the port expansion named "The embrace to Araçá Bay by land and by sea," which was supported by the researchers, environmental NGOs, and by neighboring communities that would also be affected by this economic development project.

The scientists promoted an approach with the MPA-NC advisory committee, creating in 2014 a special commission named "Araçá-Working-Group," aiming to engage representatives from the fishing sector, port authority, IBAMA, scientists and the local community. The main objective of this commission was to discuss the comprehensiveness of Araçá Bay as part of the MPA-NC delineation (Xavier et al., 2018), and later, the implementation of the ecosystem-based management plan.

Another action performed by the "Guardians of Araçá Bay" was to write a petition to the Secretary of Environment of the São Paulo State requesting modifications to the new Ecological-Economic Zoning discussed in public hearings in November 2016. Aiming to harmonize with the criteria established in other bays along the northern coast, the group requested to transform the marine area of Araçá Bay (classified as Z2M) into a Z2ME zone, a classification appropriate to fragile ecological areas. The group requested to change to a Z2M the intertidal zoning around the islets and sandy beaches (classified as Z3M), in order to allow a connection with the zoning defined in the rocky shore. Also, they requested to change the zoning in the port area (classified as Z5M) into a Z3M, with the aim of adjusting the targets to a better effluent standard. Finally, the group requested that the terrestrial range would keep the Z4 classification and not be modified to a Z5 as proposed by the government. Unfortunately, the Secretary of Environment of the São Paulo State refused all the requests of the group (Stori et al., 2019), a decision which will favor the port expansion in the future.

It is important to highlight that the Integrated Management Group elaborated many publications targeting science communication to support social learning during the whole process. These publications included the management plan itself, which contained proposals to solve 12 main problems identified in the area (Turra et al., 2016), and the publication of an illustrated book of infographics to enlighten students about the ecosystem goods and services of Araçá Bay (Xavier et al., 2017).

Another important action that helped to raise awareness about the importance of Araçá Bay was the production of a social-environmental documentary named "Pulsating: a film about Araçá Bay" (Stori et al., 2017b). The 25-min film was produced to attend another community request that emerged during the participatory meetings. The film highlights the importance of cultural maintenance, reveals the conflict with the port expansion project, and presents the social movement formed to impede the port expansion. The film-documentary spotlighted the conflict in the media and assisted strengthening of the social network.

Assessing and Applying Traditional Ecological Knowledge

Ethnoecology is the science responsible for the study of TEK, and aims to provide information about species, habitats, processes, livelihoods, and local management strategies (Marques, 2001; Toledo, 2002). Ethno-oceanography, a derivation of ethnoecology, aims to investigate the adaptive mechanisms of traditional communities regarding coastal marine environments (Moura and Diegues, 2009; Stori et al., 2012). The present work carried out ethno-oceanographic research throughout an in-depth approach, based on recurrent visits to the community to gain trust during 3 years of fieldwork (2014 – 2017).

Local practices and social mechanisms were investigated among Araçá Bay fishers and brought to discussion in the participatory planning process. All interviewees were local fishermen who used Araçá Bay as their traditional fishing territory and had a close connection with the place. Therefore,

those fishermen were considered as local experts due to their broad knowledge regarding the environment and their affinity with the area.

Recent research identified 56 people fishing in Araçá Bay, but only 26 fishers lived in the Araçá Bay area and had a close relationship with this environment (Amaral et al., 2015). A snowball sampling method (Wright and Stein, 2005) was conducted to identify the fishers that were considered experts by their peers ("*Who else do you know that has good experience about fisheries in Araçá Bay?")*. The snowball sampling method identified a total of 33 fishers. Three women were mentioned among the fishers, but they were excluded from the analysis because they were experts only in collecting clams, while the purpose was to interview fishers who had knowledge over multiple species. From the 30 fishermen mentioned, 11 were not found. Another seven fishers were found but were not interviewed because they refused or rescheduled meetings more than three times. Seventeen fishers had more than two mentions. From those, fourteen were interviewed. Another four fishers were identified *in situ*, totaling 18 interviewees. The fishers were aged between 27 and 72 years at the time of the interviews (from January to July 2015). According to these numbers, we assume that the sampling was satisfactory for reaching local experts.

In-depth semi-structured interviews (Viertler, 2002) were conducted, lasting from 2 to 3 h. All interviews were audio recorded and key information was also manually written to ensure its registration. All the interviewees gave written informed consent regarding the information provided. The questionnaire (**Supplementary Material)** focused on identifying local management practices and social mechanisms, ecosystem goods and services, changes in fisheries, disturbances promoted by economic development projects and recoveries over the decades, and future threats. The interviewees' quotes are identified in the text by their initials and age. The management practices and coupled social mechanisms identified were analyzed according to Folke et al. (1998).

A technique of ethno-mapping based on participatory mapping methods (Faria and Neto, 2006; Buarque, 2008) was also developed and applied. Ethno-mapping consists of individual drawings made by each expert during their interview, with a focus on TEK on natural resource dynamics and management practices. The information was manually drawn on an A3 size sheet previously filled with the shape of Araçá Bay coastline. Fishers were asked to point out ethno-oceanographic features such as fishery spots, habitats, preferred locations used by migratory species, type of bottom, and physical and oceanographic aspects such as tides, winds, and currents (**Figure 4**). The ethno-maps were digitized for further analysis through geoprocessing software (ArcGIS). All the interviewees gave written informed consent regarding the ethno-maps produced.

Part of the information obtained from the interviews, mainly regarding ecosystem goods and services, local problems and future threats, was organized using the SWOT framework: strengths, weaknesses, opportunities, and threats (Pickton and Wright, 1998; Srivastava et al., 2005). The SWOT is an analytical conceptual model used to group positive and negative elements that affect a certain environment or location (Buarque, 2008; Cowx et al., 2010). While strengths and weaknesses represent internal elements affecting a particular environment, opportunities and threats represent the external ones (**Figure 5**). By providing a clear organization of the elements influencing the social-ecological system, the SWOT framework can be a useful tool for integrated management and decision-making (Viegas et al., 2014).

The SWOT framework was later organized in a schematic poster and brought to participatory meetings to be completed by the participants. In order to inspire this task, the participants were encouraged to think about the ecosystem goods and services Araçá Bay provides, and to reflect upon current problems and future threats. Later, polls were conducted to classify the main results about (1) ecosystem goods and services in Araçá Bay, and (2) current problems and future threats. In Section "TEK As Support for Ecosystem-Based Management" we describe this process and discuss similarities and differences between the results obtained from TEK and the results obtained from the participatory meetings.

As previously explained, the participatory meetings engaged 141 participants from various social sectors. However, only six fishers interviewed attended the meetings. Furthermore, fishers' participation was not regular: one fisher participated in six of the seven meetings, one participated in three meetings, one participated in two meetings, and three fishers participated in only one meeting. Due to the small number of fishers in the meetings and their irregular attendance, it is not possible to affirm that the fishers strongly influenced the results of the participatory process, although their opinions were undoubtedly considered and respected by the other participants.

The area of traditional uses and activities mapped by the ethno-oceanographic research was brought to the debate about the comprehensiveness of Araçá Bay in the meetings of the Araçá Working Group (associated with the governance system of the MPA-NC).

The participatory meetings of the Local Plan for Sustainable Development and the meetings of the Araçá Working Group were monitored during the entire process to analyze how the information obtained from TEK contributed to decision-making in both forums. Participant observation was also conducted in every visit to Araçá Bay, in order to verify the social mechanisms of *caiçara* culture, to observe fisheries practices and to understand the environmental dynamics. The whole process was photo and video documented and stored in an external hard drive. The most representative images can be found in the social media of the Local Plan for Sustainable Development of Araçá Bay[1].

RESULTS

Management Practices and Social Mechanisms Based on TEK

The social-ecological system of Araçá Bay is characterized by a small-scale fishing community, which has been transformed by

[1]www.facebook.com/PLDSARACA



FIGURE 4 | An example of an ethno-map drawn by one interviewee. The ethno-map was turned 180° to facilitate reading.

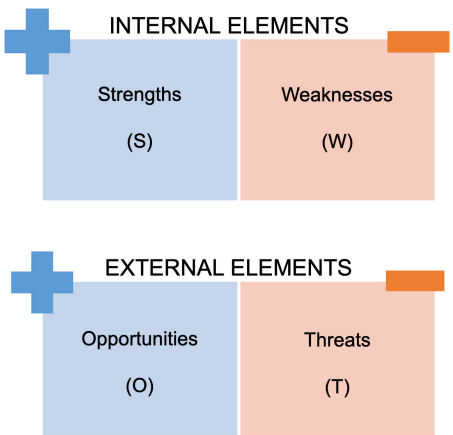


FIGURE 5 | Representation of the SWOT framework.

economic development projects associated with oil production and port expansion for almost 90 years (Peres et al., 2016). Economic growth stimulated the urban development of the region and, as a result, many fishers moved far from the shore due to the expropriation of their lands. Furthermore, the younger workforce was taken up by the oil industry and port activities. In addition to the environmental changes, economic development projects can disturb the *caiçara* culture by disconnecting the fishers from nature.

Caiçara people are a composed mix of Portuguese, Africans (ex-slaves) and indigenous people that inhabited the Southeast Brazilian coastal zone since colonial times (1500 d.c.), relying on nature for their social and material reproduction (Diegues, 1983; Begossi, 1998; Adams, 2000). It is important to highlight that the *caiçara* culture is safeguarded by the National Policy for Traditional People (PNPCT, Federal Decree N° 6.040/2007), which defines such communities as "*culturally differentiated groups and self-recognized as such, with own forms of social organization, that occupy and use the territories and the natural resources as a condition for their cultural, social, religious, ancestral and economic reproduction using knowledge, innovations, and practices generated and transmitted by tradition.*"

To verify interviewees' connections with the *caiçara* culture and its relationship with the environment, fishers were asked "What is it to be a *caiçara*?". This question revealed the social mechanisms and related aspects of *caiçara* culture, which endorsed the traditional feature of this community. All the 18 interviewees recognized themselves as *caiçara*, consistent with

the legislation statement. The fishers also linked *caiçara* culture to fisheries livelihoods, family relationships, reciprocity, and a lifestyle that "flows with the sea cycles." Folke et al. (1998) classifies these features as social mechanisms of generation, accumulation, and transmission of TEK, structure and dynamics of institutions, cultural internalization, and worldview and cultural values, as exemplified by the following statements:

> "*It is to be born and raised in a beach environment and nourish the caiçara culture. It is to fish and eat what you fished. It is to prepare the garden and picking the seasonings that you have planted.*" (S.S.O., 27).
> "*Yes, I am a native caiçara. . . Caiçara is to live in constant contact with nature, with fishing. . . I used to say that if the port constructs here, we would stay down there trapped together.*" (N.N.B., 43).

Most of the interviewees stated that they began to fish when they were between five and 10 years old (10 mentions). Three of them reported that they began before 5 years old and five interviewees reported they started fishing at 12 years of age. Fathers were mentioned as fishery mentors by 15 interviewees, but other relatives were also important to intergenerational knowledge transmission, including grandfathers (4), uncles (4), and mothers (1 mention). All the interviewees reported that they taught fisheries knowledge and skills to their children (9), friends (9), outsiders (5), and other relatives (4), but they also stated that urban development and the decrease of environmental quality were driving younger people away from fishing. An example of what Folke et al. (1998) identifies as a social mechanism for intergenerational transmission of TEK is exemplified below:

> *"Fishing is like this, my grandfather taught my father, who taught me. We learned from our ancestors that fishing in Araçá Bay should have three aspects: science, frequency, and patience. In fisheries, it happens that you go out to fish and don't catch anything. According to my knowledge, science shelters all the qualities; this is because the absence of fish may be related to the cold water or bad wind. . . Then I return on the other day and it works. Why? That's the frequency! You should go out every day. One day you catch, another day you don't catch. . . you need to 'punch the card'. . . you need patience."* (M.N.J., 72).

Regarding fishing practices, the interviewees demonstrated comprehensive TEK about 57 fishing species comprising knowledge regarding biodiversity distribution, breeding areas, and feeding areas based on habitat morphology and depth. All the 18 interviewees considered the entire bay important for fish production and, according to fishers' classifications on species habitats, the bay has seven main fishing spots: rocky shore and Araçá headland (10 mentions), islets (10 mentions), sandy bottom (9 mentions), São Sebastião channel and deep rocky reefs (6 mentions), port pier (5 mentions), small channel parallel to the beach (3 mentions) and mangrove (3 mentions).

> "*I fish on the rocky reefs near here. I mark the right places for fishing. . . I guide myself by the hills. . . I row the canoe in a direction and I crisscross the landmarks, then I find where the fishing ground is. Usually, in the rocky reefs we catch the bottom fish that eat clams. In the south direction, there are the rocky reefs and also a gravel bottom. . . that's where we catch the groupers*". (D.M.O., 62).

The ethno-maps provided information about the intensity of use and the comprehensiveness of Araçá Bay fisheries. Shallow areas of Araçá Bay (**Figure 6**), where the bay is daily exposed during the low tide, were mentioned as the locations of more intense use. This pattern was associated with habits of searching for clams, crabs and other invertebrates in shallow areas, where access is easier. The deepest areas of the bay, only accessible by boat, where most of the fish captures occur, were used with less intensity due to access difficulties. Nevertheless, during high tides, it is possible to fish by boat or canoe inside the whole area of Araçá Bay.

Fifteen interviewees affirmed that they had their own fishing boats and fourteen of them declared to share their vessels with relatives and friends when fishing. Fifteen declared that they also shared their fishing gears. Sharing is an acknowledged feature of small-scale fisheries and a recognized social mechanism of cultural value (Folke et al., 1998). All the interviewees considered the bay as an open and free access area and affirmed that the fishing spots were also shared.

All interviewees stated that there were no customary rules in Araçá Bay's fisheries. However, it was possible to identify social mechanisms of respect (seven mentions), cooperation (3), and secret (2). Yet, conflicts of use with other fishers were mentioned by four interviewees and social mechanisms of control and cultural sanctions, such as to exclude outsiders, were reported. Those social mechanisms are defined by Folke et al. (1998) as structure and dynamics of institutions and they are even more important due to the absence of a formal fishery union and/or of a communitarian association in Araçá Bay. The statements below illustrate respectively such social mechanisms:

Respect	*"If I put a fishing net and if you are not the owner, the only rule is not to touch. . . It is the respect. . . People respect, but nowadays it is not as it used to be, there are people who don't have the caiçara culture."* (D.M.O, 62).
Cooperation	*"Here there is no rivalry, on the contrary, it is camaraderie, one helps the other."* (M.N.J., 72).
Secret	*"In diving, for instance, if we catch a good fish we keep the secret about the fishing spot."* (M.A.O, 44).
Use conflicts	*"Araçá Bay is free for everyone to fish. You just should watch out for theft. . . this may happen, but they are not people from here, they are outsiders."* (I.S.F., 49).
Cultural sanctions	*"They should register the traditional fishers and exclude outsiders, allow fisheries only for the maintenance of the caiçara culture."* (E.P., 41).

The interviewees also reported practicing integrated management of multiple species and resources rotation, according to the season (summer and winter), tide, and fishing

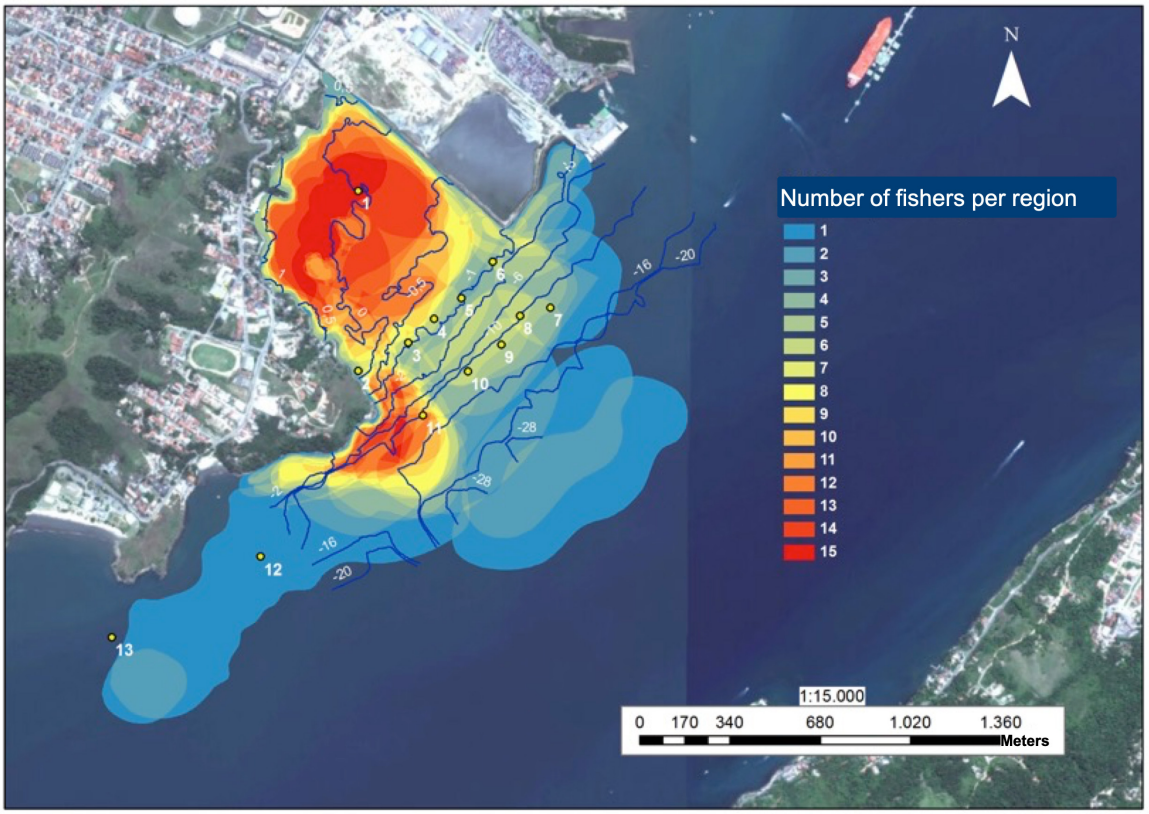


FIGURE 6 | Number of fishers per region. The blue lines correspond to depths (in meters).

spot. Such practices are recommended by Folke et al. (1998) as important strategies to build resilience in order to overcome natural disturbances. The interviewees' fishing targeted between 7 and 21 different ethno-species, with an average of 12 target species per fisher.

> *"In the summer season (mid-October to February) I fish most of the fish except the mullet that occurs in the cold season (June to July). From the edge of the channel and inside Araçá Bay, I catch the croaker and the hake with hook and line. From the Araçá Headland to the deepest part I use the fishing net to catch the 'guaiú,' sardine, 'parati'* (a little mullet), *mullet, and 'carapicu'."* (M.N.J., 72).

All 18 interviewees considered that Araçá Bay conservation is crucial, and suggested management practices to achieve it. Overall, 13 interviewees suggested restrictions on fishing. Seven fishers suggested implementing closed seasons for three ethno-species (shrimp, soft-shell crab and anchovy). Five interviewees proposed to ban fisheries in the entire Araçá Bay for recovery (3 to close the bay permanently and 2 to close it just for a period). However, eight fishers disagreed with idea of implementing no-take areas inside the bay. One fisher suggested closing only the Araçá headland (the rocky shore). Two fishers proposed excluding specific fishing techniques (trawlers and spear guns) and another two mentioned the necessity of improving fisheries enforcement. Only one fisher recommended the maintenance of total protection to an endangered group of animals, the turtle (which is already protected by law). The following statements are examples that represent the management practices identified in fishers' responses which correspond to local practices recommended to build resilience in social-ecological systems (Folke et al., 1998):

Temporal restrictions	*"In the summer there is a lot of shrimp trawling, but there is no enforcement, the closed season could last until April."* (N.N.B., 43)
Protection of vulnerable stages in the life-history of species	*"Each fisher has the awareness of not catching fish with roe and the smaller ones. When we collect the cockles, we don't catch the little ones, we let them grow. This is an awareness that came from our ancestors."* (N.N.B., 43)
Protection of specific habitats	*"If it were to create a kind of sanctuary here, I would close the marine area around 150 m from Araçá Headland until the lighthouse. Because all kinds of species live there."* (A.C., 60)
Total protection of certain species	*"I think the only rule is to release the turtle. . . we told a man to release the turtle."* (M.A.O, 44)

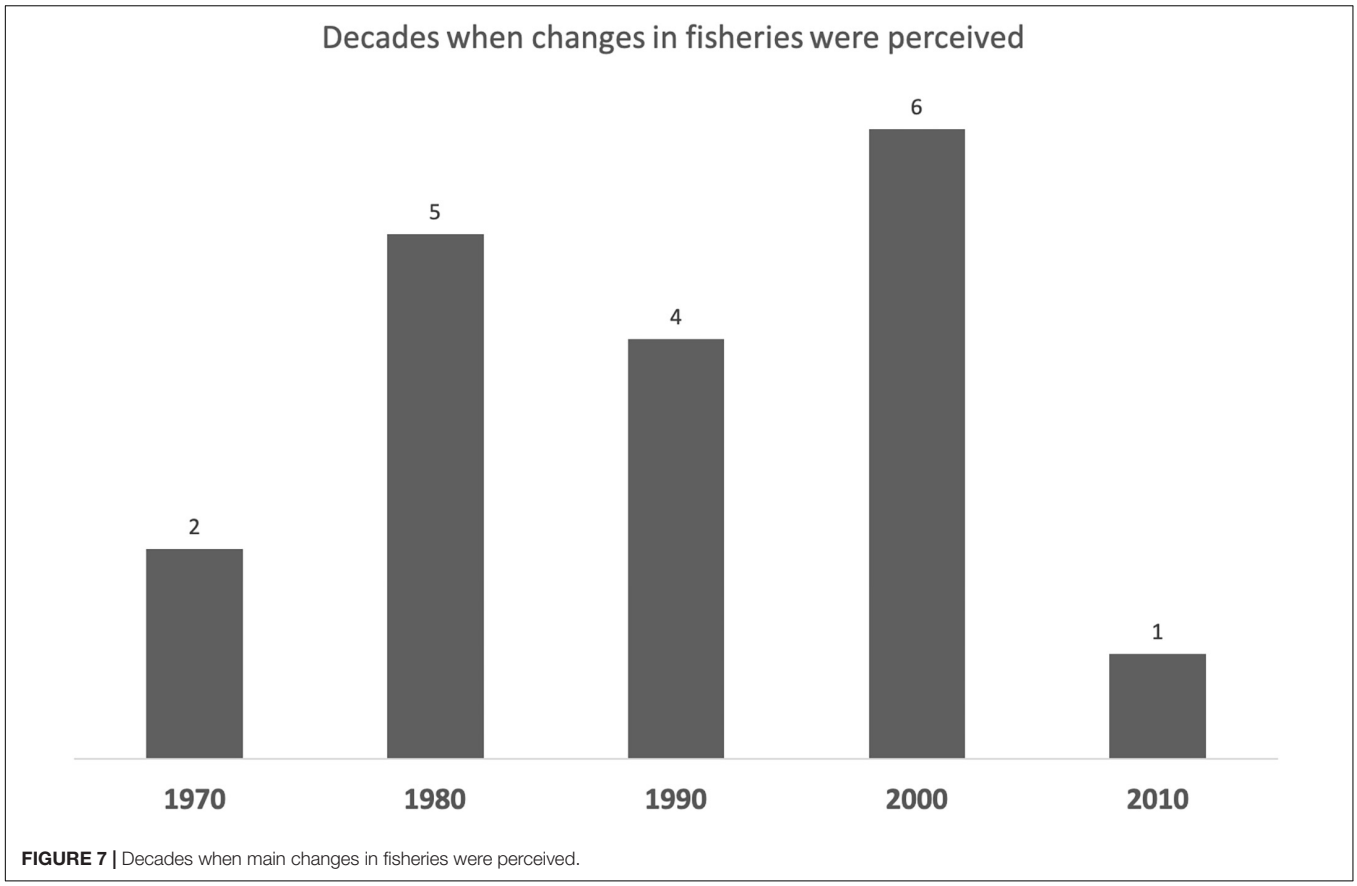


FIGURE 7 | Decades when main changes in fisheries were perceived.

Human-Induced Disturbances and Social-Ecological Consequences

All interviewees reported historical disturbances that led to the decline in fish stocks and biodiversity richness. Two fishers mentioned that they perceived major changes in fisheries since the 1970s, five fishers since the 1980s, four fishers since the 1990s, six fishers since the 2000s, and, just one interviewee observed changes since 2010 (**Figure 7**). The main reasons for fisheries decline were assigned to the port construction by land reclamation and dredging (11 mentions), port activities such as lights, noise and boat traffic (8), impacts of oil production (5), pollution by sewage and solid waste (5), irregular urban development (4), and excessive fishing outside Araçá Bay (2).

1970 *"The change here was cruel. . . there was a beach that amended with the city waterfront, it was a continuous shore, a magnificent beach! They constructed the land reclamation in such a way that they have destroyed everything. The Petrobrás destroyed a big part of the hill to build the land reclamation. They placed all the stones to make the first spit, which starts in the 'Mãe Isabel' river and ends at that place where they want to do the port expansion. The port and the Petrobrás were guilty".* (M.N.J., 72).

1980 *"About 42 years ago there was a small inlet inside the bay, where a lot of fish raised. We could fish grouper there. The Araçá was beautiful. . . We used to cross it swimming, we couldn't walk there. . . The bottom was firm sand and the water very clean. Nowadays if you walk from one island to another, you sink into the mud. Thousands of fish have died. . . What brought a lot of dirt, a lot of mud, was the dredging they did here".* (A.C., 60).

1990–2000 *"This beach had no mud, it was a sandy bottom. They dredged so the ships could dock and the mud came to halt here. Because the tide carried the mud into the bay and formed that mud bar, which should have had about 20 cm thick. So, all this area turned in to a mud about 10 years ago. Today the bay has recovered but it may happen again. Still, many residents don't take care of their sewer, people who came from other places and that occupied part of the waterfront and of the mangrove".* (I.S.F., 49).

The interviewees also reported a partial recovery in the system, which led to a new state in the ecosystem balance. They also described adaptive management practices, which corresponds to the literature recommendations toward SER (Folke et al., 1998): the monitoring of changes in the ecosystem and in resource abundance, responding to and managing pulses and surprises, and nurturing sources

of ecosystem renewal. The statements below demonstrate fishers' management practices based on TEK, and the adaptive management strategies adopted to face a new ecosystem state:

> *"The fisheries have declined a lot, several species have disappeared because of these attacks that have happened... the bass, the hake, the shrimp has diminished, the mangrove has disappeared, crabs have disappeared, some birds too, because it is the food chain... The first attack was due to the port land reclamation, a lot of mud came from the port. The second attack was the sewage pipeline and the third attack was another sewage pipeline of SABESP* (water company) *that affected everything there... Because of these changes a mud has formed, so today we fish more these mud species: 'piragica,' 'parati,' 'betara,' because they feed more in the mud. But in the past the focus was on the big fish: hake, sea bass, whiting, but they are practically extinct with these changes."* (W.S.B., 45).
>
> *"We lost in quantity, but the bay is very resistant! This mangrove is very strong... the mangrove says: I won't die! It survives even with oil spills because it has a natural washing, it has a very strong flow of water."* (E.P., 41).
>
> *"The sand changed a lot, it was firmer. It turned into mud and we couldn't walk there anymore. Now it's getting better. The fisheries changed a lot, but now it is returning."* (S.R.J., 37).
>
> *"In addition to the regattas and beach cleanings that we organize, I talk to the residents to raise environmental awareness. I also teach kayaking and soccer; my students are my great hope. I try to show them the importance of Araçá Bay. I want to register our NGO to have the means and the strength to speak on behalf of the community, to actually represent the residents and to be able to search for support."* (N.N.B., 43).

Fifteen interviewees considered that no other areas of Araçá Bay should be taken for port expansion. Three interviewees considered that the port could be expanded only toward the boundary between the current port and Araçá Bay, because this area is already impacted and "dirty" due to the port activities. However, any port expansion over this location would lead to a new dirty and impacted area.

> *"I think that if the port expansion really happens the way they're saying, will be the end! Because it will really mess up the ecosystem. They said they won't construct a land reclamation anymore, now will be on a slab. But most of the fish species don't like shade... It will affect the algae, turtles, everything, and Araçá Bay is a breeding area for many species."* (M.A.O., 44).

Gathering all the information above, nine local practices based on TEK recommended by Folke et al. (1998) were identified in Araçá Bay's small-scale fishery system: (1) monitoring change in ecosystems and in resource abundance; (2) total protection of certain species; (3) protection of vulnerable stages in the life-history of species; (4) protection of specific habitats; (5) temporal restrictions on harvest; (6) multiple species and integrated management; (7) resource rotation; (8) responding to and managing pulses and surprises; and (9) nurturing sources of ecosystem renewal. In addition, four types of social mechanisms classified by Folke et al. (1998) as important in fostering SER were identified: (1) generation, accumulation, and intergenerational transmission of TEK; (2) structure and dynamics of institutions (role of stewards or wise people, and social and cultural sanctions); (3) mechanisms for cultural internalization; and (4) worldview and cultural values such as sharing, reciprocity, and respect.

Ecosystem Goods and Services, Problems and Threats Identified in TEK

Detailed information obtained from the entire fishers' interview analysis was identified and organized through the SWOT framework (**Figure 8**). The elements identified, either positive or negative and internal or external, were grouped into four different clusters (strengths, weaknesses, opportunities, and threats). While ecosystem goods and services (e.g., fisheries, biodiversity and touristic potential) were classified as strengths and opportunities, current problems and uncertainties (e.g., oil spills, sewage pollution, and port expansion) were classified as weaknesses and threats.

When asked specifically "*What is good in Araçá Bay?*", and "*Why is Araçá Bay important?*" fishers mentioned various ecosystem goods and services, which were arranged in seven main groups: fisheries, food provision and income source (15 mentions); affinity with the place and *caiçara* culture heritage (12 mentions); breeding area and biodiversity maintenance (8 mentions); tranquility and life quality (7 mentions); nature, sea breeze and mangroves as protection from coastal erosion (7 mentions); leisure activities and tourism (7 mentions); sheltered area, good for navigation and to moor fishing boats (3 mentions).

Further, when specifically asked "*What is not good in Araçá Bay?*", fishers reported several problems that were arranged in eight main groups: solid waste pollution (9 mentions); sewage pollution (8 mentions); port operation and expansion activities (7 mentions); presence of drug users in Araçá Bay surroundings (6 mentions); oil pollution (4 mentions); dredging activities (4 mentions); urban and industrial growth (2 mentions); and absence of local government (2 mentions). The same questions were asked in the participatory meetings, and the results are presented in the next section.

TEK as Support for Ecosystem-Based Management

The information based on TEK contributed to support decision-making in two ways: in the participatory meetings of the Local Plan for Sustainable Development of Araçá Bay, and in the

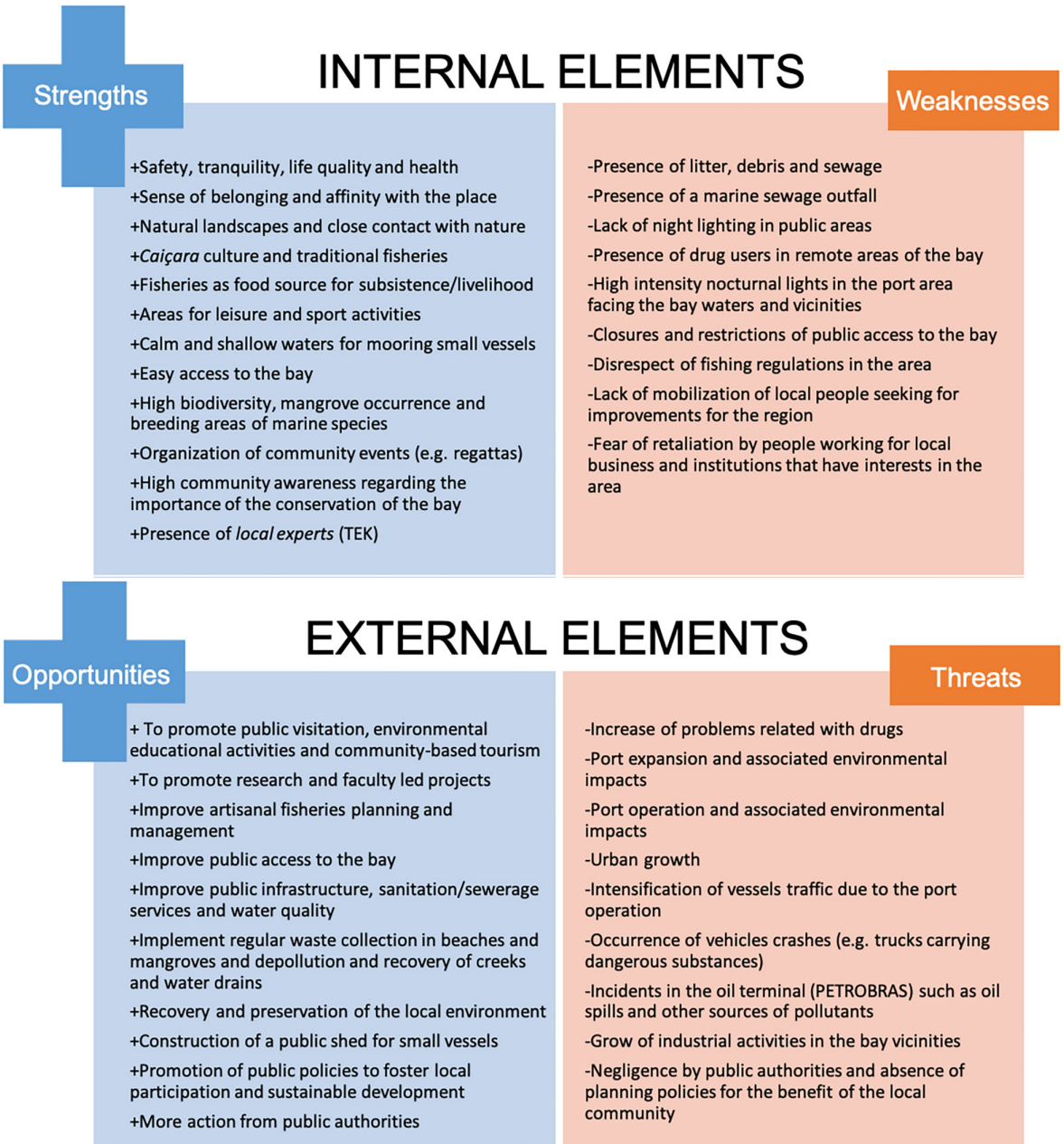


FIGURE 8 | SWOT framework detailing ecosystem goods and services, problems and future threats found in fishers' interviews.

Araçá Working Group of the Marine Protected Area of the Northern Coast (MPA-NC).

TEK as Support to the Local Plan for Sustainable Development of Araçá Bay

During the participatory meetings the participants were encouraged to identify potentialities and fragilities, considering current and future projections, by answering the following questions: "*What is good in Araçá today and for tomorrow?*"; and, "*What is not good in Araçá today and for tomorrow?*". The ecosystem goods and services identified were combined with the research results of the Biota-Araçá Project (Turra et al., 2016; Carrilho and Sinisgalli, 2018), and with the results obtained from TEK through the SWOT framework. The combination of all this information was organized in a schematic poster presented to the participants to be completed during the meetings (**Figure 9**).

In order to prioritize the ecosystem goods and services organized in the schematic poster the participants were asked to think about "*What Araçá Bay does for us?*". A total of 20 ecosystem goods and services were grouped and participants were asked to vote on the most important ones. The results were: food provision, small-scale fisheries, and source of

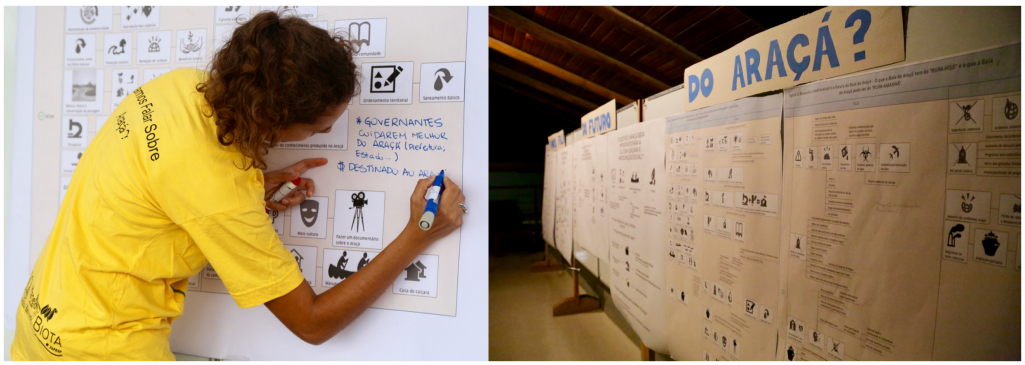


FIGURE 9 | The information obtained from SWOT was turned into icons and presented in posters to be completed during participatory meetings.

income (15 votes); education and research (14 votes); *Caiçara* culture maintenance (12 votes); social participation (11 votes); biodiversity maintenance (9 votes); supply of raw material for handcrafts (6 votes); leisure, recreation, sport and tourism (5 votes); species breeding (5 votes); future benefits (4 votes); easy access to the sea (3 votes); mangrove occurrence (3 votes); satisfaction with environmental conservation (3 votes); coastal protection (3 votes); affinity to the place (2 votes); diversity of landscapes (1 vote); scenic beauty and landscape conservation (1 vote); man-nature contact area (1 vote); effluent treatment (1 vote); and carbon storage (1 vote). The item "sheltered area for mooring boats" (previously identified in the debates) did not receive any vote, and the item "health and quality of life" was included in the list after the participatory discussion and the conclusion of poll results. It is important to highlight that food provision through small-scale fisheries, including its importance as a source of income, and linked social mechanisms of *caiçara* traditional culture were acknowledge by the participants as fundamental ecosystem goods and services in Araçá Bay.

The correspondence between the information obtained from TEK and the information obtained during the participatory meetings is demonstrated in **Figure 10**. All the information TEK provided was considered and classified by the participants in a similar order of importance to that given by the fishers. Nevertheless, distinct ecosystem goods and services were considered by the participants of the meetings: education and research; social participation; supply of raw material for handcrafts; future benefits; satisfaction with environmental conservation; coastal protection; diversity of landscapes; scenic beauty and landscape conservation; effluent treatment; and carbon storage. These results do not mean that the fishers do not recognize those ecosystem benefits, but perhaps that the questions applied in the ethno-oceanographic interviews were not sufficiently precise, or that more elucidation about the questions might have been necessary. Additionally, when analyzing the SWOT framework, it is possible to verify that other strengths and opportunities based on TEK were considered in the participatory process (such as the potential for developing tourism, health and quality of life, and sheltered area for mooring boats), because the SWOT framework was indeed totally incorporated into the schematic poster. The SWOT framework, in turn, was not able to provide an ordering of those features because it was based on the entire interviewees' responses. Only a classification based on well-defined questions could provide a rigorous method for ordering fishers' opinions. Due to this limitation, the classification was performed based on only two questions of the questionnaire ("*What is good in Araçá Bay?*", and "*Why is Araçá Bay important?*"). Nevertheless, ecosystem services linked to regulation processes and future benefits appeared to be too subtle to be identified by the fishers (such as effluent treatment, carbon storage and coastal protection). The process of social learning stimulated by the participatory meetings proved to be effective for the acknowledgment of such subtle ecosystem goods and services.

A poll regarding the current problems and future threats was also undertaken. Participants were asked to vote on the most concerning problems, considering a list of 12 main problems, which were identified and grouped in previous meetings. The 12 main problems were: sewage pollution (19 votes); the current port and Petrobrás activities (19 votes); solid waste pollution (17 votes); urban and industrial growth (16 votes); chemical pollution (15 votes); social problems associated with drug users (12 votes); inefficiency in management (5 votes); illegal fishing or overfishing (2 votes); impacts on mangroves (1 vote); current port structure (1 vote); low social control (1 vote); and infrastructure for leisure and tourism (1 vote).

Similar to the results regarding ecosystem goods and services, all the information provided by TEK was assimilated and acknowledged by the participants of the meetings (**Figure 11**). However, the order of importance was slightly different in this case. Additionally, other problems and threats were classified by the stakeholders, such as illegal fishing and overfishing, impacts on mangroves, low social control, and lack of infrastructure for leisure and tourism. The results demonstrate that fishers perceive the problems that directly affect fisheries. On the other hand, the stakeholders were stimulated to think about the causes that led to these problems, and in doing so they identified problems that affect the social-ecological system as a whole, including other economies (such as tourism) and the deep causes of environmental

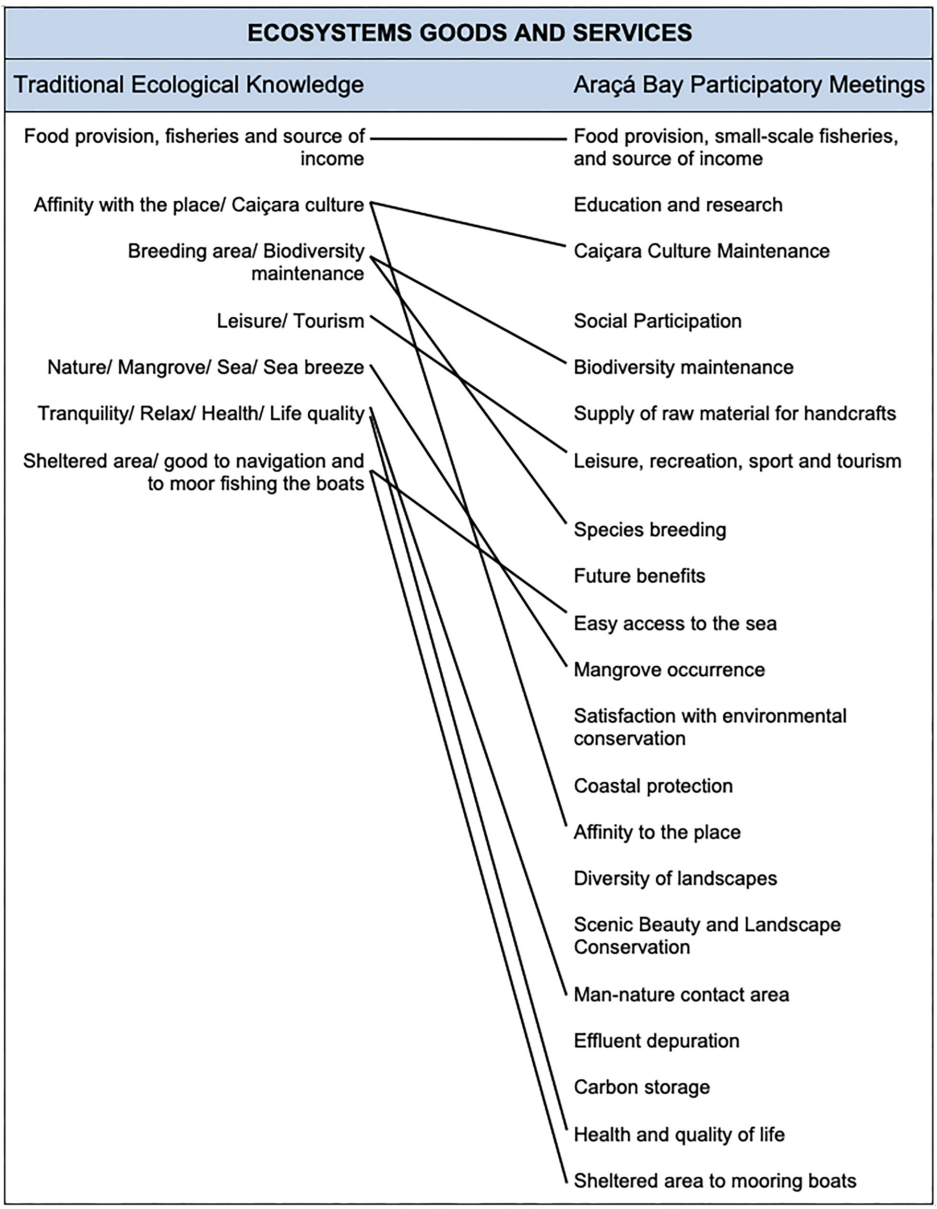


FIGURE 10 | Correspondence between the ecosystem goods and services classified by fishers and those classified by the participants.

problems (such as the low social control). Illegal fishing and overfishing were also identified as threats by the participants. However, the fishers did not consider them as main problems, although excessive fishing outside Araçá Bay was mentioned by them when questioned about historical changes in fisheries. Similar to the results regarding ecosystem goods and services, the ordering of current problems and future threats was possible only using one specific question of the questionnaire ("*What is not good in Araçá Bay?*"). Other weaknesses and threats classified by the participants can, however, also be identified in the SWOT framework, such as lack of mobilization of local people toward improvements for the region and the negligence of public authorities (which can be related to the low social control), and non-compliance with fisheries regulations.

TEK as Support to the Araçá Working Group of the Marine Protected Area of the Northern Coast (MPA-NC)

TEK was used to inform the Araçá Working Group of MPA-NC in order to discuss the comprehensiveness of Araçá Bay in MPA design. Traditional uses and activities identified from TEK were grouped with other scientific information produced by the Biota-Araçá Project (e.g., bathymetry, type of bottom, sand dispersion, marine currents, larvae dispersal, benthic species distribution, legislation), to provide information

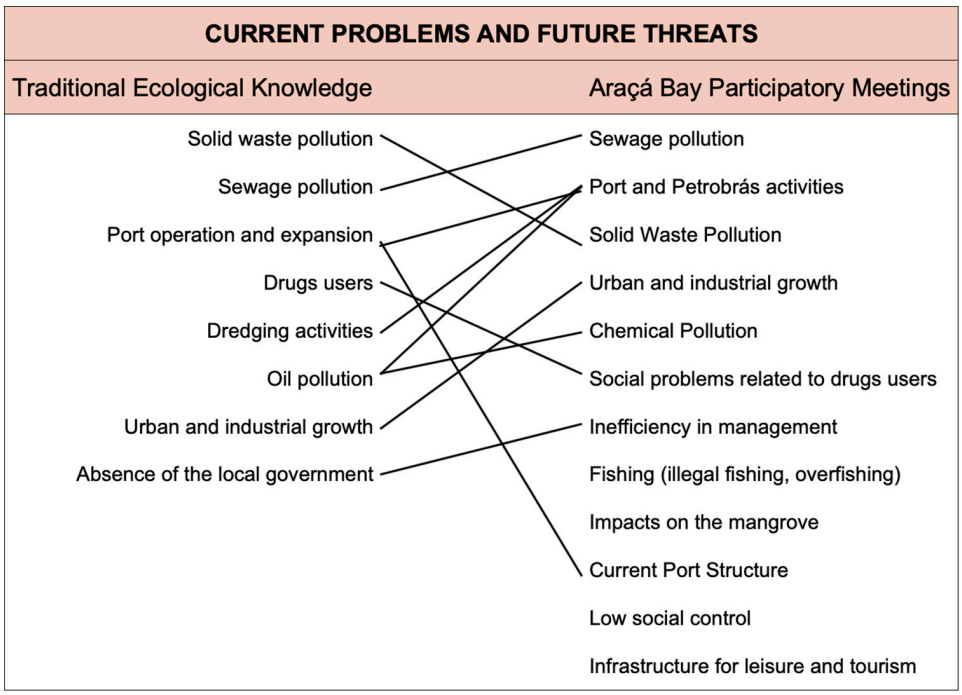


FIGURE 11 | Correspondence between the main problems and future threats classified by fishers and those classified by the participants.

about Araçá Bay and assist the decision-making process (Xavier et al., 2017).

The area of traditional uses and activities indicated by the fishers was chosen as the main criterion to define the comprehensiveness of Araçá Bay within the MPA-NC (**Figure 6**). Some fine adjustments were made to the map and the final proposal was submitted to the MPA-NC advisory committee to be discussed during the elaboration of the MPA-NC management plan (which is expected to be concluded in 2019). The comprehensiveness of Araçá Bay's small-scale fisheries territory was verified and the participatory planning process endorsed its significance as a management unit.

DISCUSSION

The Role of TEK to Inform Participatory Ecosystem-Based Management

Araçá Bay is an example of how human-induced disturbances can lead to continuous shifts in ecosystem state, affecting the whole social-ecological system. Complex adaptive ecosystems tend to have multiple stable states, or stability domains, toward which they progress and organize (Colding et al., 2003). Such shifts can occur in nature but tend to be exacerbated by human activities that simplify ecosystems and often cause loss of biological diversity and ecosystem services (Nyström et al., 2000). Hence, human-induced disturbances can shift an ecosystem to a less desirable functional state or to an irreversible one (Colding et al., 2003).

Fishers interviewed demonstrated their knowledge of past human-induced disturbance and recognized the consequences of an imminent threat to the coastal marine environment, such as the port expansion project that would shift this ecosystem to an irreversible state. Combined with the development of infrastructure projects planned for the region, such as roads and oil industry, the port expansion could intensify the disturbances in Araçá Bay by eliminating ecosystem structure and functions, impacting management practices and linked social mechanisms, and reducing social capital and the ability to adapt to environmental transformations.

The Araçá Bay social-ecological system evidenced important local practices and social mechanisms advocated by Folke et al. (1998) as essential features to provide flexibility in natural resources management, and to adapt to changes. The capacity to diversify and adapt fisheries according to resource availability, as verified in the system analyzed, is an important strategy to overcome natural disturbances and can be an advantage to face human-induced disturbances up to a tolerable level. It is likely that the high variety of environments combined with the high tidal amplitude have enabled the settlement of a high diversity of organisms adapted to these environments. Consequently, fishers have developed a vast array of fishing techniques and adjusted them to diverse environmental and oceanographic conditions. Therefore, the environmental diversity might have favored the diversity of local practices and social mechanisms found in this particular social-ecological system, and consequently, its adaptive capacity.

Management practices and social mechanisms are demonstrated to manage natural disturbances. However, these features alone were not able to constrain successive human-induced disturbances due by economic development projects in Araçá Bay, including oil production, port construction, and urban growth. In this case, the fishers were able to use their accumulated knowledge only to adapt fisheries according to these changes, not to influence these changes. The knowledge acquired regarding human-induced impacts and disturbances should be considered in further environmental licensing assessments, such as port expansion assessments, and in management plans at local and broader scales.

Some researchers have suggested that the flexibility of customary management systems has made them resilient to population growth and economic modernization. Others suggested that these systems might be effective common-pool resource-management institutions in situations of relatively low population density and subsistence economies, but could die out in response to factors such as increased population pressure, commercialization of resources and the breakdown of customary authority (Cinner and Aswani, 2007). Management practices and social mechanisms of the *caiçara* culture preserved in Araçá Bay might be powerful features to inform decision-making and help to design a sustainable future in this social-ecological system, despite the cultural erosion provoked by the urban and industrial growth.

Casimirri (2003) argued that fundamental issues about how TEK is defined are at the root of the barriers to incorporation of traditional values and knowledge into contemporary management practice. That author emphasized that there is a need to move beyond the current discourse in which TEK is merely a form of data to 're-define' TEK in resource management. The main question, according to that author, is how management systems and the TEK that informs them can form the basis of community-based, adaptive institutions of resource management. The question should not be so much "How to integrate TEK?" but 'How to integrate TEK holders into resource management?'. Where TEK holders have direct involvement in management processes through community-based, adaptive resource decision-making institutions, there is a much greater potential to meaningfully incorporate TEK into sustainable resource management (Casimirri, 2003).

When fostering a participatory planning process and striving to integrate TEK holders in this process, the researchers of the Biota-Fapesp/Araçá Project struggled to break the inertia of the community regarding the "lack of mobilization of local people toward improvements for the region," identified by TEK holders and participants. The researchers acted as facilitators to engage fishers and other stakeholders and endeavored to codify all the information obtained and set a common language, in a process known as "translation."

Processes of translation are recognized strategies to promote concertation in social networks, encouraging collective decision-making processes (Callon, 1986; Beuret, 2006; Beuret et al., 2006). The translation of the importance of ecosystem goods and services, and of the causes of current problems and future threats, was fostered by the researchers in the participatory planning process, providing bases for the Local Plan for Sustainable Development of Araçá Bay (the ecosystem-based management plan), and for the Araçá Working Group of the MPA-NC. TEK proved to be an important source of information to support decision-making in both processes, and was recognized and incorporated by the participants of these forums, helping them to achieve the desired outcomes (**Figure 12**).

Berkes et al. (2007) analyzed integrated management in the Canadian North, assessing its contribution to the advancement of knowledge and practice regarding the role of indigenous knowledge and community-based monitoring. The authors confirmed the relevance of TEK and stakeholders' participation to widen the range of knowledge in order to understand and help monitor environmental change.

Mutually beneficial outcomes in participatory research, to both indigenous/local communities and resource management agencies, have been widely documented in the marine resource management literature. However, participation might not always be of interest to target communities, and some forms of participation can actually be coercive (Shackeroff and Campbell, 2007). In advocating that researchers consider participatory engagements in conservation research, these authors encourage a critical treatment of the concept toward a true collaboration rather than superficial forms of participation. Partnerships in which all sectors gain from the participatory meetings are possible, given a situation in which potential power dynamics, ethical issues, and cultural context are explored, articulated and respected conscientiously throughout the research process. Only through an informed and conscientious approach can TEK be incorporated into conservation research in a manner beneficial to both conservation and TEK holders, achieving biological and socio-economic goals in a culturally appropriate manner, and recognizing and respecting TEK (Shackeroff and Campbell, 2007).

Further, Wiber et al. (2004) suggested that participatory research should target the needs of local communities by identifying interests among resources users, and designing and carrying out research projects to meet these needs. Their research demonstrated the effectiveness of extending participatory methods to challenge traditional scientific notions of the research process (Wiber et al., 2004).

Bélisle et al. (2018) analyzed 23 published studies regarding the integration of scientific ecological knowledge and TEK into ecological modeling and found that participatory research is a helpful tool to reach the full potential of combining both forms of knowledge. However, methodological guidelines are not completely settled yet, especially regarding participatory methods, and the most pressing challenge relies in the integration of methods and concepts from the social and natural sciences (Bélisle et al., 2018).

Ruiz-Mallén and Corbera (2013) performed a broad literature review with the aim of investigating how TEK, community-based conservation, and SER interrelate. The authors found that, in co-management initiatives, local people also benefited from cross-institutional arrangements and scientific knowledge that contributed to capacity building, knowledge generation through mutual learning, and trust building. Only by fully

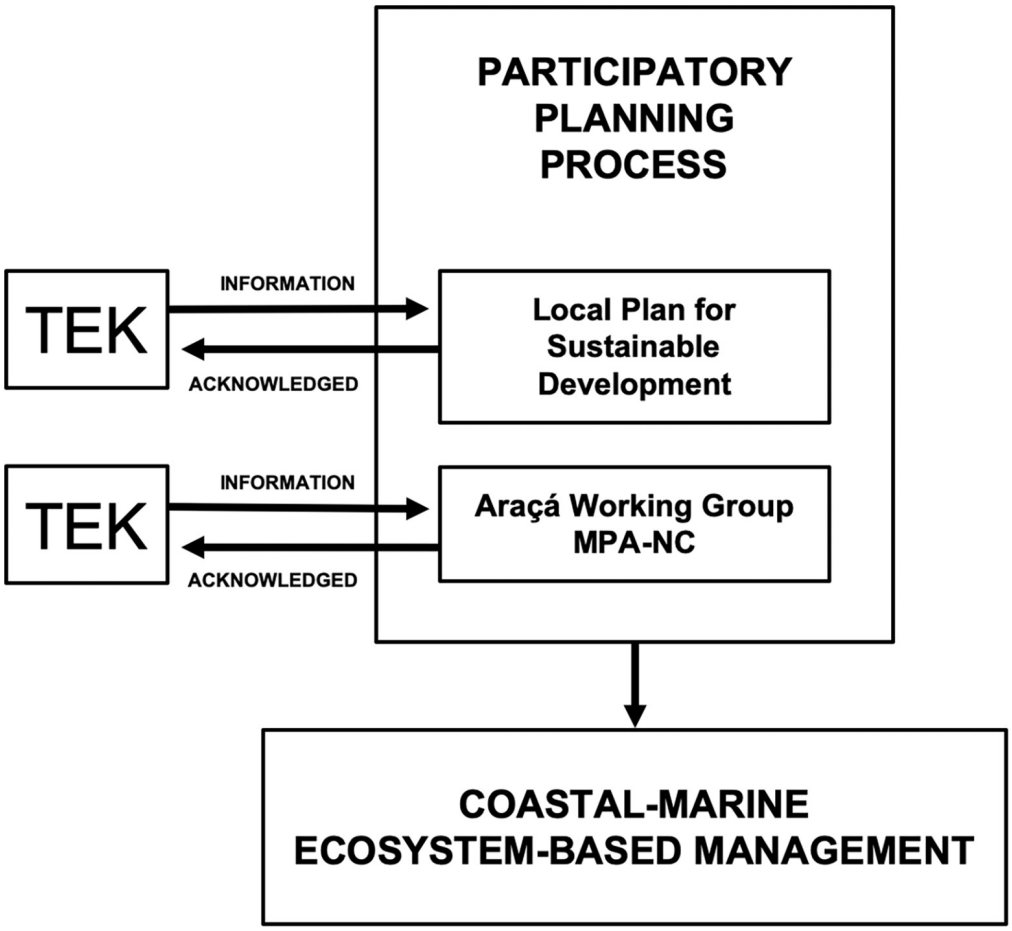


FIGURE 12 | Scheme of the application of TEK to support ecosystem-based management.

comprehending existing synergies, conflicts, and trade-offs between TEK, community-based conservation, and adaptive capacity in changing environments will it be possible to understand the complexities of social-ecological systems and guide decision making for conservation across governance scales in meaningful ways (Ruiz-Mallén and Corbera, 2013).

Combining scientific and TEK stands as a promising approach to design strategies that are both scientifically sound and attuned to local value systems and priorities (Gómez-Baggethun et al., 2013). TEK systems are increasingly acknowledged for their contribution to sustaining biodiversity and ecosystem services, as well as being important reservoirs of experiential knowledge that can provide significant insights for the design of adaptation and mitigation strategies to build SER in the face of global environmental change (Gómez-Baggethun et al., 2013).

Cinner and Aswani (2007) suggested that scientists and policy makers should encourage and strengthen institutional hybrids between customary and modern management systems. Hybrid institutions offer considerable potential for sustainable resource management by harnessing TEK, respect for traditions, scientific knowledge, and local acceptance. Hybrid management should understand and harness both scientific and local knowledge systems and mechanisms for detecting and reacting to changes in social-ecological systems. The participatory process during the establishment of hybrid strategies is critical for capturing TEK and for explaining scientific knowledge (Cinner and Aswani, 2007).

The emergent hybrid management initiated by the Biota-Fapesp/Araçá Project was successful in breaking the *status quo* of continuous human-induced disturbances in Araçá Bay, by creating a powerful resistance against the port expansion project. The port expansion and coupled economic development projects will be constant threats to this social-ecological system. Therefore, the established stakeholders' network supporting Araçá Bay conservation must keep connected and vigilant. Furthermore, governmental institutions, including the municipality, the port authority, and the Marine Protected Area, should engage with the community in discussions about the planning and management of the bay, acknowledging TEK as an important source of knowledge which, associated with scientific knowledge, can help to design a better future for this coastal marine area.

Science should provide key guidance in taking steps toward coastal marine ecosystem-based management and, by building

management from a foundation of the best available knowledge, the ecosystems and the services they provide can be managed or restored in relatively predictable ways (UNEP, 2011). Considering that, in some cases, TEK is the only source of knowledge available, investigating TEK and integrating TEK holders is highly recommended in order to develop ecosystem-based management plans in coastal marine areas. As the present research demonstrated, TEK can provide information about biological communities, ecological functions, oceanographic processes, natural resources management practices, and detailed information concerning the impacts of economic development projects on ecosystems. In this context, TEK played a key role in providing unique information to a participatory planning process aimed at coastal marine ecosystem-based management, in the absence of long-term scientific data.

CONCLUSION

This study has demonstrated the relevance of TEK in supporting a participatory planning process toward ecosystem-based management in coastal marine ecosystems. The effort of interviewing fishers contributed greatly to building the foundations of the participatory meetings. The process of translation in participatory meetings, combining TEK, scientific, and stakeholders' knowledge, was revealed as a powerful arrangement to foster social learning and favor SER in social-ecological systems.

At the beginning, Biota-Fapesp/Araçá Project researchers encountered some resistance from fishers and other stakeholders in participating in the research. Scientists found a disunited and powerless community due to the impact of successive human-induced disturbances caused by economic development projects. TEK research evidenced the lack of social organization in this place (a fact corroborated by fishers and participants of the meetings), a characteristic that undermines SER in social-ecological systems. However, throughout the project's implementation, the researchers gained the community's trust and people gradually joined the participatory meetings, culminating in the creation of a group of citizens concerned about implementing the plan and being aware of the conservation of Araçá Bay. As a result of this enthusiastic participatory planning process, the court canceled the environmental license for the port expansion, a rare event for environmental licensing in Brazil. We conclude that, in situations of weakened communities, external aid can be crucial in rebuilding the confidence to claim collective rights.

TEK analysis proved to be an important approach to provide environmental data in a human-induced disturbed ecosystem and to support planning in this coastal marine social-ecological system. There is a global need for a better understanding of SER in human-induced disturbed systems. Social-ecological research should advance the understanding of how humans interact with nature, how resilient these systems are in dealing with external and internal crises, and how to better inform decision-making toward sustainability. In order to support participatory ecosystem-based management strategies toward SER and sustainability of coastal marine social-ecological systems, researchers and practitioners should consider traditional territories in planning, recognize local management practices and linked social mechanisms, and elicit TEK on ecosystem goods and services and on the impact of human-induced disturbances.

ETHICS STATEMENT

This study was carried out in accordance with the recommendations of the Ethics Committee on Animal Experimentation of Oceanographic Institute of the University of São Paulo. The Ethics Committee on Animal Experimentation of Oceanographic Institute of the University of São Paulo exempts from formal ethical approval, studies employing non-invasive research methods with humans and which do not benefit from the utilization of the biodiversity genetic resources, in line with national legislation. All subjects gave written informed consent in accordance with the Declaration of Helsinki.

AUTHOR CONTRIBUTIONS

FS, CP, and AT designed the research. FS and CP collected the data and performed the data analysis. CP transcribed the interviews and produced the maps. FS, CP, AT, and RP wrote the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00571/full#supplementary-material

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Gaps in Protection of Important Ocean Areas: A Spatial Meta-Analysis of Ten Global Mapping Initiatives

Natasha J. Gownaris[1], Christine M. Santora[2], John B. Davis[3] and Ellen K. Pikitch[2]*

[1] Department of Environmental Studies, Gettysburg College, Gettysburg, PA, United States, [2] School of Marine and Atmospheric Sciences, Institute for Ocean Conservation Science, Stony Brook University, Stony Brook, NY, United States, [3] OCTO (Open Communications for The Ocean), Woodinville, WA, United States

To safeguard biodiversity effectively, marine protected areas (MPAs) should be sited using the best available science. There are numerous ongoing United Nations and non-governmental initiatives to map globally important marine areas. The criteria used by these initiatives vary, resulting in contradictions in the areas identified as important. Our analysis is the first to overlay these initiatives, quantify consensus, and conduct gap analyses at the global scale. We found that 55% of the ocean has been identified as important by one or more initiatives, and that individual areas have been identified by as many as seven overlapping initiatives. Using our overlay map and data on current MPA coverage, we highlight gaps in protection of important areas of the ocean. We considered any area identified by two to four initiatives to be of moderate consensus. Over 14% of the ocean fell under this category and most of this area (88%) is not yet protected. The largest concentrations of medium-consensus areas without protection were found in the Caribbean Sea, Madagascar and the southern tip of Africa, the Mediterranean Sea, and the Coral Triangle. Areas of high consensus (identified by five to seven initiatives) were almost always within MPAs, but their no-take status was often unreported. We found that nearly every marine province and nearly every exclusive economic zone contained area that has been identified as important but is not yet protected. Much of the identified area lies within contiguous stretches of $>100{,}000$ km^2; it is unrealistic to expect that all this area be protected. Nonetheless, our results on areas of consensus provide initial insight into opportunities for further ocean protection.

Keywords: areas beyond national jurisdiction, Aichi Target 11, marine protected area, overlay analysis, hotspots, representativeness, Sustainable Development Goal 14, World Database on Protected Areas

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****Correspondence:***
Natasha J. Gownaris
ngownaris@gmail.com

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INTRODUCTION

There is currently a convergence of global interest in ocean science and conservation. Some examples of this interest include the United Nations (UN) declaration of 2021–2030 as the "Decade of Ocean Science for Sustainable Development"; the convening of five global, high-level *Our Oceans* conferences; the adoption of the first UN Sustainable Development Goal devoted exclusively to ocean issues (SDG 14); and the focus on implementation of the Convention on Biological Diversity's (CBD) Aichi Targets. Both the SDG and CBD targets (specifically, SDG 14.5 and Aichi Target 11) call for protection of at least 10% of the ocean by the year 2020.

Scientists have argued that adequately protecting biodiversity and meeting socioeconomic goals is likely to require a much higher level of protection than 10%, with estimates that 30–50% of the ocean should be set aside in Marine Protected Areas (MPAs) (IUCN, 2016: WCC-2016-Res-050-EN; O'Leary et al., 2016; Wilson, 2016). Discussions are underway to determine the next set of targets and activities under CBD's Post-2020 Biodiversity Framework, including those for MPAs. All signs point to the likelihood that targets for 2021 and beyond will exceed the current agreed targets for 2020 (Campbell and Gray, 2019).

Scientific evidence that MPAs can help to maintain and restore fish populations (Sala and Giakoumi, 2017; Aalto et al., 2019), increase ecosystem resilience (Mellin et al., 2016; Roberts et al., 2017), and provide socio-economic benefits (Bennett and Dearden, 2012; Ban et al., 2017; Rodríguez-Rodríguez et al., 2019) continues to build. These benefits are only realized, however, if MPAs are appropriately sited, strongly protected, and effectively managed (Watson et al., 2016; Ban et al., 2017; Gill et al., 2017; Giakoumi et al., 2018; Jantke et al., 2018; Rees et al., 2018; Sala et al., 2018). Many of the world's MPAs fail to meet quality standards (Barnes et al., 2018; Sala et al., 2018; Zupan et al., 2018; da Silva, 2019) and MPA siting has not always been based on the best available science (Jantke et al., 2018; Fischer et al., 2019). The United Nations Environment World Conservation Monitoring Centre estimated that 7.59% of the ocean was covered by protected areas as of March 2019 (UNEP-WCMC and IUCN, 2019a). However, only 4.8% of the ocean is in implemented and actively managed MPA and only 2.2% is in strongly protected MPAs or no-take marine reserves (Marine Conservation Institute, 2019b).

There are numerous ongoing UN and non-governmental (NGO) initiatives to map important marine regions, each of which involves extensive research and expert opinion (**Supplementary Appendix S3**: **Supplementary Table S1**). The value of several of these initiatives as roadmaps for future protection has been explicitly recognized by the CBD (e.g., CBD, 2008: UNEP/CBD/COP/DEC/IX/20; CBD, 2016: UNEP/CBD/SBSTTA/20/INF/43). Here, we present a spatial meta-analysis of ten UN and NGO initiatives that vary in purpose, scope, and selection criteria. As noted by similar initiatives in terrestrial systems, differences in criteria and scope among these marine initiatives may result in contradictory maps and lack of a clear message about which regions should be prioritized (Mace et al., 2000; Soutullo et al., 2008). Our overlay is a direct response to discussions among scientists, MPA practitioners, and diplomats during and following a 2016 workshop in Rome, where policymakers expressed uncertainty over which map should be used as a starting point for future protection (**Supplementary Appendix S1: Rome Call to Action**). By overlaying maps from ten of the major global initiatives, we seek to highlight areas that are consistently identified as important despite differences in the criteria used.

There have been several spatial meta-analyses focused on initiatives identifying important terrestrial regions (Brooks et al., 2006; Soutullo et al., 2008; Schmitt et al., 2009; Iwamura et al., 2013). The criteria used to identify important terrestrial and marine regions have also been extensively discussed elsewhere (Brooks et al., 2006; Gilman et al., 2011; Marchese, 2015; Briscoe et al., 2016). However, the identification of important marine areas has lagged similar initiatives for terrestrial areas (Briscoe et al., 2016) and our analysis is the first to overlay these initiatives, quantify consensus, and conduct gap analyses at the global scale. We identify gaps in protection of previously identified important areas and examine the strength of protection where it does exist. We also consider gaps in protection of regions identified as important in the context of biogeographic representativeness. Our aim is to provide a starting point for future protection by focusing attention on areas of the ocean that have been identified as important but that remain unprotected.

MATERIALS AND METHODS

Initiatives

To be considered in our analysis, initiatives had to: (1) be conducted under the auspices of a UN agency or an NGO; (2) identify important marine regions at the global scale; and (3) have spatial data available online. Application of these criteria resulted in selection of 10 initiatives that varied in purpose, scope, and methodology (**Table 1**). Among those initially considered, Mission Blue Hope Spots was the only global initiative that we did not include in the final analysis, as the boundaries of Hope Spots were not sufficiently defined for use in spatial analysis.

In each initiative we collated, the number of sites identified as important ranged from under 20 to over 200, with a total area per initiative ranging from approximately 60 thousand km^2 to over 80 million km^2 (**Supplementary Table S1**). Some initiatives included both terrestrial and marine regions (e.g., World Heritage Sites, WHS), while others included only marine regions (e.g., Vulnerable Marine Ecosystems, VMEs). A few initiatives focused on (e.g., VMEs) or included (e.g., Ecologically or Biologically Significant Marine Areas, EBSAs) areas beyond national jurisdiction (ABNJ), while others were based on proposals by member states and, to date, have focused solely on exclusive economic zones (EEZs) (e.g., Particularly Sensitive Sea Areas, PSSAs: Roberts et al., 2010; Fletcher et al., 2017; WHS: Laffoley and Freestone, 2017).

The motivation for identifying marine areas of importance also varied greatly by initiative. Myers et al. (2000), for example, originally suggested that the long-standing Biodiversity Hotspots (updated in Mittermeier et al., 2011), be used to prioritize use of limited conservation funds. Though identified based on terrestrial biodiversity, the boundaries of these regions also enclose important marine areas such as coral diversity hotspots (Roberts et al., 2002) and coastal marshes (Myers, 2003), and Conservation International has established marine programs within these regions. By comparison, MARPOL Special Areas were not established to prioritize conservation funds, but to impose special restrictions on ship pollution in ecologically vulnerable ocean areas (International Maritime Organization [IMO], 2002). We use the term "important" to refer to identified regions, acknowledging that what qualifies as important varies with initiative.

TABLE 1 | Criteria used by ten initiatives to map important marine areas.

	UN initiatives						NGO initiatives			
Criterion*	**EBSA**	**PSSA**	**RAMSAR**	**MARPOL**	**VME**	**WHS**	**CI**	**IBA**	**WWF**	**AZE**
Uniqueness	■	■	■		■		■			
Threatened species	■	■	■		■	■		■	■	■
Life stage importance	■	■	■		■			■		
Vulnerability	■	■		■	■		■		■	
Productivity	■	■	■		■					
Biodiversity	■	■	■			■	■			
Naturalness	■	■	■			■			■	
Structure					■					
Geomorphological importance		■				■				
Currently impacted		■					■			
Human dependency		■				■				
History/Heritage		■				■				
Research		■				■			■	
Importance to species/subspecies			■					■		

EBSAs, Convention on Biological Diversity, Ecologically or Biologically Significant Marine Areas; MARPOL. International Maritime Organization, MARPOL Special Areas; PSSAs, International Maritime Organization, Particularly Sensitive Sea Areas; Ramsar, Convention on Wetlands, Ramsar Sites; WHS, Educational, Scientific and Cultural Organization, World Heritage Sites; VMEs, Food and Agriculture Organization, Vulnerable Marine Ecosystems; Hotspots, Conservation International, Biodiversity Hotspots; IBAs, Birdlife International, Important Bird and Biodiversity Areas; AZE, Alliance for Zero Extinction, Zero Extinction Sites; WWF, World Wildlife Fund, Marine Priority Areas.
See **Supplementary Appendix S2: Initiative Criteria for more details.*

Spatial data for most initiatives examined in this study were available online as occurrence polygons, through either organizational websites or the ArcGIS data portal (**Supplementary Appendix S3**: **Supplementary Table S1**). For EBSAs, we compiled a complete spatial database by merging the 275 individual EBSA files available on the CBD website as of October 2017. For databases that had not been updated for several years (e.g., Alliance for Zero Extinction sites, AZE), we contacted the relevant organization to verify that updated data were not available prior to analysis. We thoroughly reviewed each initiative's available documentation to determine which criteria were used when important marine areas were identified and how these criteria were defined (**Supplementary Appendix S2: Initiative Criteria**).

Additionally, we collated academic studies that aimed to identify important marine regions at a global scale (e.g., Selig et al., 2014; Klein et al., 2015; Martin et al., 2015; Jenkins and Van Houtan, 2016; Ramírez et al., 2017; Fischer et al., 2019) but ultimately did not include them in our analysis. Academic papers typically present results for a snapshot of time, while most UN and NGO initiatives are ongoing. Furthermore, while our analysis relied on occurrence polygons, many academic studies produce raster data (e.g., geomorphic feature diversity: Fischer et al., 2019). Because raster data are continuous, defining occurrence polygons based on these data would have required us to decide subjectively which raster values constituted important regions, which would have been beyond the scope of our study.

Overlay Analysis

We conducted all analyses in ArcGIS Desktop 10.5 (ESRI, 2016). To allow for consistent global calculations of area, we first projected all spatial data to Eckert IV. For global-level maps, Eckert IV is the equal-area projection system with the lowest weighted mean error of scale distortion (Canters, 2002; Šavrič et al., 2016; Jenny et al., 2017). We clipped all initiative layers to the Natural Earth 10 m Ocean polygon prior to analysis (Ocean Version 4.1.0: Natural Earth, 2018b) to ensure that they had the same spatial extent.

To conduct our overlay analysis, we merged all ten initiatives into one layer ("merged polygons layer") and identified regions of overlap using a modification of the Count Overlapping Polygons tool (Honeycutt, 2012). Our application of this tool was as follows: (1) Create a new layer (Feature to Polygon tool) with separate polygons showing either single initiatives or regions where initiatives intersected ("overlap polygons layer"); (2) Calculate the centroid for each overlap polygon (Feature to Point tool); (3) Spatially join (one-to-one join) the centroid data to the merged polygons layer and count the number of polygons overlapping with each centroid; and (4) Join the centroid count data back to the overlap polygons layer.

The result of this overlay was a map consisting of over 134,000 occurrence polygons ("overlay map"), each including information on the number of initiatives ("overlapping initiatives") identifying that specific region as important. Hereafter, we refer to areas identified by one or more initiatives as "identified areas." Within that, we refer to areas identified by two to four initiatives as "moderate-consensus areas" and by five to seven initiatives as "high-consensus areas."

Because the initiatives we included varied substantially in the size and number of regions identified, we conducted a sensitivity analysis. We repeated the steps outlined above ten times, removing one initiative prior to each run and recalculating the area identified at each level of overlap.

We also tested for interdependence of initiatives, i.e., one initiative identifying an area based solely on identification by a prior initiative. For example, spatial data on Important Bird Areas (IBSs) were considered at EBSA workshops (Bax et al., 2016), which were also attended by members of several conservation organizations that have produced global maps of important marine areas, including Birdlife International and World Wildlife Fund (Johnson et al., 2018). To examine the degree of interdependence, we iteratively clipped each initiative by each other initiative and calculated pairwise percent overlap.

Gap Analysis

Current Marine Protected Area Coverage

To identify gaps in protection of important marine regions, we considered our overlay map in the context of current global MPA coverage (**Figure 1**). There are two databases commonly used to track global MPA coverage, MPAtlas (Marine Conservation Institute, 2019b) and the World Database on Protected Areas (WDPA; UNEP-WCMC and IUCN, 2019a). The WDPA is a joint project between UN Environment (UNEP) and the International Union for Conservation of Nature (IUCN). MPAtlas builds on WDPA, providing additional fact-checking on reported statistics and additional information on MPA status and protection level (Marine Conservation Institute, 2019b).

We used the marine dataset of the January 2019 World Database on Protected Areas (WDPA; UNEP-WCMC and IUCN, 2019a) to conduct our gap analysis because the WDPA is the official entity to which all member states report progress on protected area coverage under both CBD Aichi Target 11 and Sustainable Development Goal 14.5. Following the WDPA's methodology for calculation of current coverage (UNEP-WCMC and IUCN, 2018a) and other recent academic studies (Jantke et al., 2018; Johnson et al., 2018; O'Leary et al., 2018), we did not include UNESCO Man and Biosphere Reserves or MPAs with a status of "proposed" or "not reported" as currently protected in our analysis. In most analyses, except for when information on specific MPAs was required (e.g., level of protection), we dissolved the WDPA prior to analysis to prevent double counting of overlapping polygons (UNEP-WCMC and IUCN, 2018b).

Size

Using a meta-analysis of 87 global MPAs, Edgar et al. (2014) found that sites covering at least 100 km^2 were more effective in conserving biodiversity than those that were smaller. Given the relationship between MPA size and effectiveness, we calculated and considered the size distribution of identified areas without current protection, using the dissolve tool to combine contiguous areas (**Figure 1**). Exploratory analysis of the resulting data suggested that our analyses resulted in sliver polygons, defined by Delafontaine et al. (2009) as "a polygon, resulting from the combination of at least two different geographical layers, of which the boundary is supposed to coincide with a line, but does not because of position errors and/or uncertainties." In our data, for example, one of the identified marine areas is Port Davey, Australia, which is part of the larger Tasmanian Wilderness WHS. However, clipping the WHS layer to the ocean layer also resulted in 670 polygon "slivers" less than 10 km^2 in size, 634 of which were less than 0.01 km^2 in size.

Sliver polygons are common in spatial analyses that combine datasets created by different organizations and can sometimes be identified based on geometric properties such as area, perimeter-to-area ratio, and thinness ratio (Delafontaine et al., 2009). Preliminary analysis of our data, however, suggested that none of these properties could be used to systematically identify sliver polygons. Area and perimeter to area ratio both had a strong positive skew across polygons and were normally distributed when log-transformed, so did not indicate thresholds with which to define polygons as slivers. Thinness ratio showed a uniform distribution, so similarly did not provide any clear thresholds for identifying slivers. We therefore report results for all polygons in our study. Sliver polygons represent a very small fraction of the total unprotected area of identified regions; less than 0.02% of the total area, for example, was found in polygons of less than 10 km^2.

Exclusive Economic Zones

Although protecting marine ABNJ is of growing interest (e.g., UN General Assembly Resolution 72/249; Heffernan, 2018; United Nations General Assembly, 2018; Wright et al., 2018), most MPA creation continues to take place within EEZs (UNEP-WCMC and IUCN, 2019a). Immediate marine protection is likely to be most feasible at the national level, as global negotiations on a legally binding treaty to protect marine biodiversity in international waters will conclude in 2020 at the earliest (High Seas Alliance, 2019). We therefore focused part of our gap analysis on EEZs (Flanders Marine Institute, 2018; **Figure 1**). Following Fischer et al. (2019), we excluded disputed and joint regimes from our analysis and considered all Antarctic territories as one EEZ managed by the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR), resulting in a total of 230 EEZ polygons. We clipped the resulting EEZ layer by a layer representing all identified areas, protected identified areas, and unprotected identified areas, respectively, to calculate the area-weighted average overlap, percent identified area protected, and total unprotected identified area.

Biogeographic Representativeness

Ecological representativeness of MPAs is explicitly called for in Aichi Target 11 and remains a challenge in the global MPA network (Jantke et al., 2018). We examined gaps in biogeographic representativeness using spatial data on marine provinces as defined by the Marine Ecoregions of the World (MEOWs: Spalding et al., 2007) and the Pelagic Provinces of the World (PPOWs: Spalding et al., 2012). These are complementary, non-overlapping classifications; MEOWs and PPOWs represent coastal regions (<200 m depth contour) and pelagic regions (>200 m depth contour), respectively. Nearly all MEOW area is located within EEZs, but EEZs also contain PPOWs. We used MEOW's 62 provinces and PPOW's 37 provinces as our biogeographic units (**Figure 1**). As with EEZs, we calculated the area-weighted average overlap, percent identified

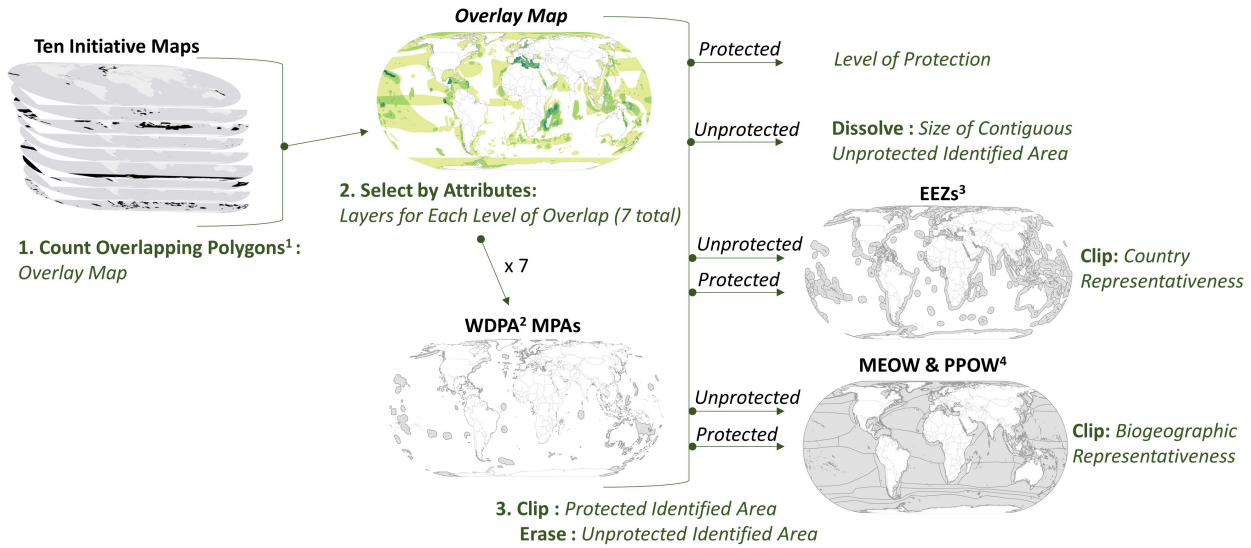


FIGURE 1 | Conceptual diagram showing spatial analysis workflow. Names of data layers resulting from our analysis are italicized. (1) Count Overlapping Polygons Tool (Honeycutt, 2012), (2) World Database on Protected Areas (UNEP-WCMC and IUCN, 2019a), (3) Exclusive Economic Zones (Flanders Marine Institute, 2018), (4) Marine Ecoregions of the World (Spalding et al., 2007) and Pelagic Provinces of the World (Spalding et al., 2012).

area protected, and total unprotected identified area for each MEOW and PPOW province.

RESULTS

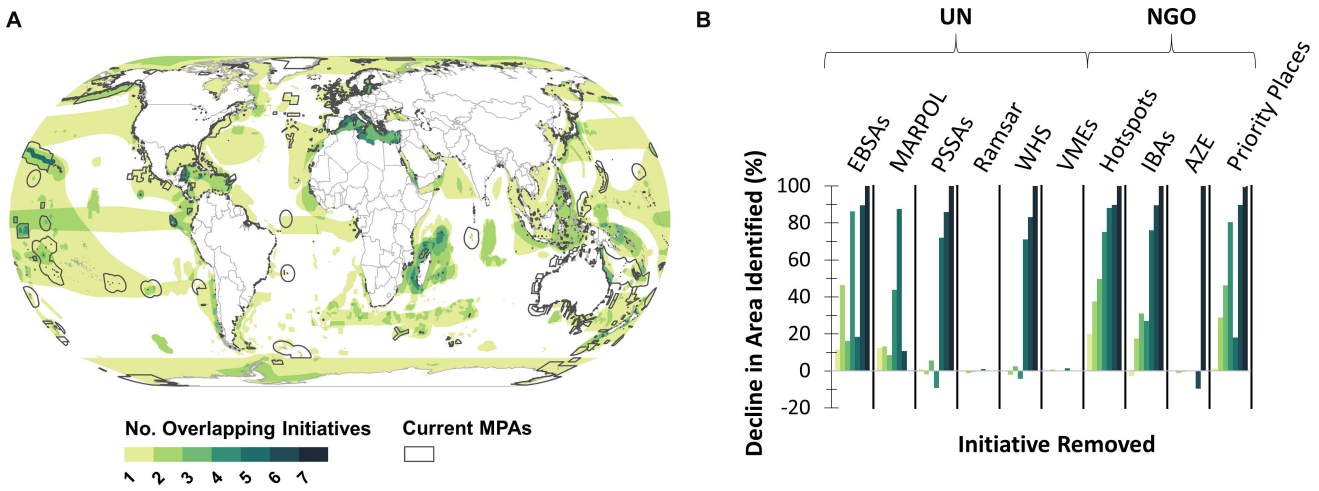
Overlay Analysis

Approximately 55% of the global ocean was identified as important by one or more UN and/or NGO initiative(s). Within this 55%, the number of overlapping initiatives ranged from one to as many as seven, with the following area covered by each: 40% of the ocean by one initiative, 12% by two initiatives, 2% by three initiatives, 0.7% by four initiatives, and <0.5% by five or more initiatives (**Figure 2**). Some high-consensus areas were small and distinct, such as the ocean surrounding the Galápagos Islands. This was the only area with seven overlapping initiatives, the highest-observed level of overlap. Large areas of moderate- and high-consensus also occurred in the Caribbean Sea, Madagascar and the southern tip of Africa, the Mediterranean Sea, and the Coral Triangle (**Figure 2**).

Initiatives varied substantially in the number of criteria used to identify important areas, from one criterion for some initiatives (AZE Zero Extinction Sites: presence of threatened/endangered species; MARPOL Special Areas: system vulnerability/fragility/sensitivity) to 12 criteria for PSSAs (**Table 1** and **Supplementary Appendix S2: Initiative Criteria**). However, within an initiative, not all criteria were necessarily met for a region to be considered important. For example, some EBSAs were identified based only on one "critically important" criterion (Johnson et al., 2018). The most commonly used criterion across initiatives was the presence of threatened or endangered species, used by seven of ten initiatives. Other commonly used criteria (each used by five or six initiatives) included naturalness, biodiversity, vulnerability, uniqueness/rarity, and importance to life stages (**Table 1** and **Supplementary Appendix S2: Initiative Criteria**).

When considered as two categories, the area identified by at least one NGO initiative or by at least one UN initiative covered 37 and 28%, respectively, of the ocean (**Supplementary Appendix S3**: **Supplementary Figure S1**). However, the overlap between areas identified by at least one UN initiative and areas identified by at least one NGO initiative was low (**Supplementary Appendix S3**: **Supplementary Figure S1**). The low overlap can be attributed, in part, to differences in how UN and NGO initiatives identified important areas. While UN initiatives relied on an average of 6.5 criteria (range 1–12), NGO initiatives relied on an average of three criteria (range 1–4). There were also differences in which criteria were most often used to identify important areas; for example, while four UN initiatives included the criterion of "naturalness," only one NGO initiative did (WWF Priority Areas) (**Table 1** and **Supplementary Appendix S2: Initiative Criteria**).

Among the initiatives examined, areas identified by the CBD initiative (EBSAs) had the greatest total marine extent (**Table 1**). Our sensitivity analysis showed that removing EBSAs and Hotspots led to the largest decline in total area identified, with declines of 20 and 26%, respectively (**Figure 2**). Half of the initiatives had little (<1% of total area) impact on the area identified (VMEs, PSSAs, WHS, Ramsar Sites, AZE Sites). The areas identified by these initiatives were small and often overlapped with those identified by several other initiatives, i.e., were found in high-consensus areas (**Supplementary Appendix S3**: **Supplementary Table S2**). Removing PSSAs, WHS, Ramsar sites, or AZE sites therefore led to declines of between 72.6 and 88.5% in high-consensus area (**Figure 2**).



EBSAs: Convention on Biological Diversity, Ecologically or Biologically Significant Marine Areas
MARPOL: International Maritime Organization, MARPOL Special Areas
PSSAs: International Maritime Organization, Particularly Sensitive Sea Areas
Ramsar: Convention on Wetlands, Ramsar Sites
WHS: Educational, Scientific and Cultural Organization, World Heritage Sites
VMEs: Food and Agriculture Organization, Vulnerable Marine Ecosystems
Hotspots: Conservation International, Biodiversity Hotspots
IBAs: Birdlife International, Important Bird and Biodiversity Areas
AZE: Alliance for Zero Extinction, Zero Extinction Sites
WWF: World Wildlife Fund, Marine Priority Places

FIGURE 2 | Results of overlay analysis for marine areas identified as important by six United Nations and four non-governmental organization initiatives. The number of overlapping initiatives in each region is mapped in **(A)** and the impact of each initiative on the area identified at each level of overlap is shown in **(B)**. Initiatives identified 55% of the ocean as important, with up to seven initiatives overlapping. Overlap map **(A)** also shows current marine protected area coverage (World Database on Protected Areas, January 2019: UNEP-WCMC and IUCN, 2019a); most areas identified as important are not currently protected, except for areas identified by at least five initiatives **(A)**. Areas with overlap of at least two initiatives are concentrated in the Caribbean, the Mediterranean, the Coral Triangle, Madagascar and surrounding islands, and the Galápagos and surrounding islands **(A)**. The two initiatives with the largest impact on the area were Convention on Biological Diversity EBSAs and Conservation International Hotspots **(B)**. Countries are displayed using data provided by Natural Earth (Admin 0 Countries 4.1.0; Natural Earth, 2018a).

The average pairwise overlap in areas identified by two initiatives was 14.2% (range of 0.00–69.30% across pairs; **Supplementary Appendix S3**: **Supplementary Table S2**). WHS had the highest and most variable overlap with areas identified by other initiatives (30.9% overlap across pairs); these sites had no overlap with VMEs but a 69.30% overlap with Hotspots (**Supplementary Appendix S3**: **Supplementary Table S2**). MARPOL Special Areas showed the lowest overlap with areas identified by other initiatives (average of 5.57%, range of 0.00–19.24% overlap across pairs), overlapping by more than 6% with only Hotspots and Priority Places (**Supplementary Appendix S3**: **Supplementary Table S2**).

Gap Analysis

Current Protected Area Coverage

A total of 7.75% of the area identified as important by a single initiative was protected, 11.7% of moderate-consensus area (two to four overlapping initiatives) was protected, and 94.7% of the high-consensus area (five to seven overlapping initiatives) was protected (**Figure 3**). Here and throughout our results and discussion, we use the term "protected" to refer to all MPAs included in our analysis, regardless of their level of protection. Approximately 66% of current MPA coverage overlaps with areas identified by the initiatives examined in this analysis (**Figure 2**).

Of the identified area that was protected, 18.4% fell within MPAs listed by the WDPA as "all no-take" and 20.3% was listed as "part no-take" (i.e., having a combination of fishing and non-fishing zones). Therefore, approximately 39% of identified area that was protected was within MPAs with at least some no-take area (**Figure 3**). Most of the remaining identified area that was protected (57.1%) was within MPAs with an unreported no-take status; only 4.2% fell within MPAs that were reported as having no no-take area. Although areas of high-consensus were most likely to be protected, they were least likely to have their no-take status reported (**Figure 3**).

Size

Over 99.9% of the total area identified by the initiatives we examined was within continuous areas of at least 100 km^2, for a total of 341 areas above the size threshold defined by Edgar et al. (2014) (**Supplementary Appendix S3**: **Supplementary Figure S2**). Of these large areas, 40 covered greater than 100,000 km^2 (**Supplementary Appendix S3**: **Supplementary Figure S2**) and could be further examined as potential sites for new, very large-scale MPAs (O'Leary et al., 2018) and/or networks of MPAs (Green et al., 2015). The five largest contiguous areas were located in the Pacific Ocean (two areas each greater than 34 million km^2), in the Arctic (over 17.9 million km^2), in the Antarctic (over 19.3 million km^2),

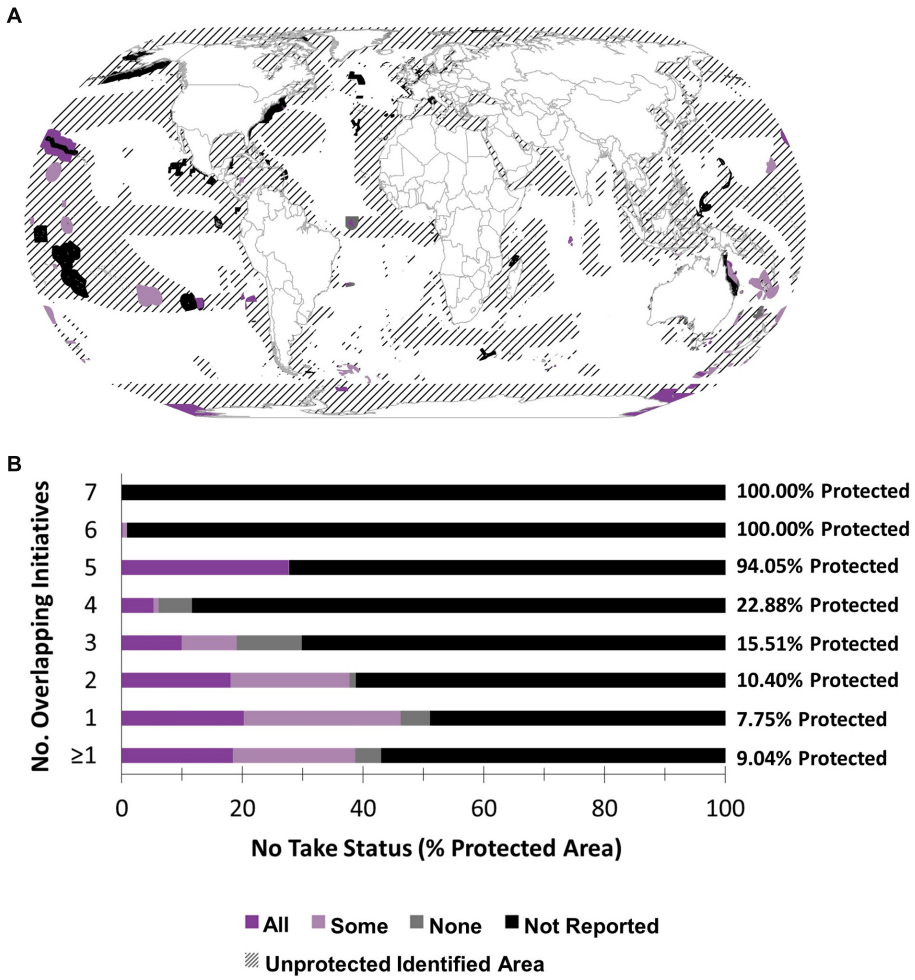


FIGURE 3 | Map showing the protection level of areas identified as important based on the World Database on Protected Areas (UNEP-WCMC and IUCN, 2019b). Identified areas outside of marine protected areas (MPAs) are shown using hatching and areas within MPAs are colored based on their no-take status **(A)**. Areas were identified as important by six United Nations and four non-governmental organization initiatives that vary in purpose and scope. The percent protection at each level of overlap and the no-take status of this protection is shown in **(B)**. As the number of overlapping initiatives in an area increased, so did the likelihood that the area was already within an MPA **(B)**. However, only 18.4% of all identified area that was protected fell within no-take MPAs and the no-take status of many protected identified areas was unknown **(A,B)**. Countries are displayed using data provided by Natural Earth (Admin 0 Countries 4.1.0; Natural Earth, 2018a).

and in the Mediterranean Sea/northwest coast of Africa (12.6 million km^2).

Exclusive Economic Zones

We found that approximately 58% of the total area identified was located within EEZs, and that over 97% of the moderate- to high-consensus area was located within EEZs. While 42% of the total area identified was located within ABNJ, <10% of the protected identified area was within ABNJ. These protected areas fell within just eight MPAs.

Of particular relevance to UN SDG 14.5, 198 of 230 EEZs had at least 10% of their waters within an area that was identified as important by at least one initiative but not currently protected (**Figure 4**). Just two EEZs had no identified area (Kazakhstan and Turkmenistan). Only 58 EEZs had at least 10% of their identified area protected, and 28 EEZs had 0% of their identified area protected. In contrast, 13 EEZs protected 98% or more of their identified area.

The EEZs with the largest area identified as important and not yet protected were French Polynesia (4.7 million km^2), Indonesia (5.1 million km^2), Canada (5.7 million km^2), and Russia (6.2 million km^2), as well as parts of the 200 nm zone around the continent of Antarctica (7.5 million km^2).

Biogeographic Representativeness

Area was identified within every MEOW and PPOW province and had greater coverage in MEOWs (average of 62.8%, range of 0.23–100.00% cover across provinces) than in PPOWs (average of 57.5%, range of 0.44–100.00% cover across provinces). However, because PPOW provinces cover a much larger total area (91.5% of the ocean) than do MEOW provinces, most of the identified area fell within PPOW provinces (86.7% of the total area).

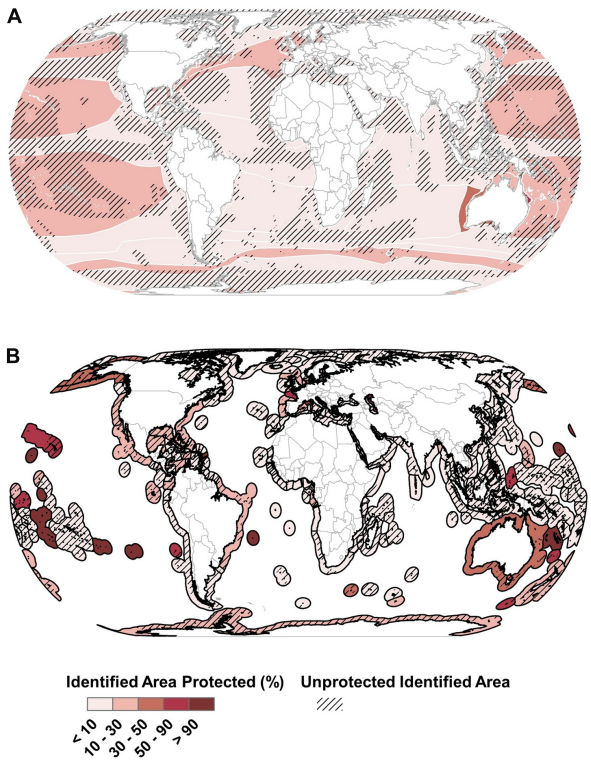


FIGURE 4 | Map showing what percent of area identified as important is currently within a marine protected area by **(A)** marine province (MEOWs, Marine Ecoregions of the World: Spalding et al., 2007; PPOWs, Pelagic Provinces of the World: Spalding et al., 2012) and **(B)** Exclusive Economic Zone (EEZs, Exclusive Economic Zones: Flanders Marine Institute, 2018). Areas were identified as important by six United Nations and four non-governmental organization initiatives that vary in purpose and scope. The boundaries of provinces in **(A)** are outlined in white and of EEZs in **(B)** are outlined in black; due to the spatial scale of the map, most MEOWs are not discernable. Representativeness, measured as the percent of identified area that was protected within each EEZ or MEOW/PPOW, was low: fewer than a quarter of EEZs and a half of marine provinces had at least 10% of their identified area protected (lightest shade of pink). Every marine province and nearly every EEZ has areas identified as important but not currently protected (hatched area). Countries are displayed using data provided by Natural Earth (Admin 0 Countries 4.1.0; Natural Earth, 2018a).

The average area-weighted overlap in identified areas was less than two was less than two for 86 provinces and was three or greater for four provinces: the Mediterranean (PPOW), the Mediterranean Sea (MEOW), Hawaii (MEOW), and the Galápagos (MEOW). Areas identified within MEOW provinces had a greater average overlap (1.63, range of 1.00–6.02 initiatives across provinces) than did those within PPOW provinces (1.37, range of 1.00–3.40 initiatives across provinces).

Areas identified within MEOWs were more likely to be protected (13.1% of total area protected; average of 21.9%, range of 0.00–100.00% across provinces) than those within PPOWs (8.1% of total area protected; average of 6.66%, range of 0.00–49.11% across provinces). Over 50% of the total protected identified area fell within just four provinces, all of which were PPOW provinces (in decreasing order of area: South Central Pacific Gyre, North Central Pacific Gyre, Antarctic, Non-gyral Southwest Pacific); 25% of the total protected identified area was within just one province, the South Central Pacific Gyre.

The only province with 100% protection of the identified area was the Galápagos (MEOW). In contrast, 60 provinces had <10% of their identified area protected, and 16 provinces had <1% of their identified area protected, including seven provinces for which no identified area was protected (PPOW provinces: Black Sea, Benguela Current, Guinea Current, Malvinas Current, North Pacific Transitional, Somali Current; MEOW province: Marquesas) (**Figure 4**).

DISCUSSION

The past decade has been marked by a growing recognition of the need for enhanced protection of marine ecosystems, including the setting of numerical targets for ocean protection by the CBD Aichi Targets and UN Sustainable Development Goals. Positive and measurable action toward meeting such targets has been made through commitments at global venues such as the Our Oceans Conferences, the UN General Assembly, the CBD Conference of the Parties, and the UN Ocean Conference. A recent study quantified the impact of MPA commitments made at the first four Our Ocean Conferences, showing that over five million km^2, or 1.4% more of the ocean, is now in implemented MPAs as a result (Grorud-Colvert et al., 2019).

Despite this progress, concerns have been raised regarding the "perverse outcomes" (Barnes et al., 2018) that can arise from numerical targets. These concerns include how MPA sites are chosen (e.g., protecting areas of low biodiversity value for political expediency) and how they are managed, or not, once they are designated (Watson et al., 2016; Jantke et al., 2018; Rees et al., 2018; Fischer et al., 2019). To meet global conservation goals, protection must be "based on the best available scientific information" (SDG 14.5) and placed in "areas of particular importance for biodiversity and ecosystem services" (Aichi Target 11).

We leveraged the extensive effort that UN and NGO organizations have put into identifying important areas of the ocean to provide a starting point for locating future MPA sites. While the ten initiatives we considered varied greatly in purpose and scope, they do show commonalities. At least five initiatives relied on one or more of the following to identify important areas: presence of threatened species, naturalness, high biodiversity, vulnerability, and/or irreplaceability. These criteria are also commonly used by initiatives to identify important terrestrial areas (e.g., Brooks et al., 2006; Gilman et al., 2011). In general, these initiatives were developed to prioritize limited conservation funding (e.g., Marchese, 2015; Briscoe et al., 2016), though not necessarily to recommend sites for future protected area placement. For example, although originally motivated by a need to identify potential ABNJ MPAs, EBSAs are explicitly described as a "scientific exercise that should not be conflated with any potential management requirements" (Dunn et al., 2014). Regardless, the criteria used by these initiatives highlight regions of high ecological value and/or vulnerability and therefore may inform the MPA selection process (Diz et al., 2018).

Our overlay of these initiatives showed that at least 49% percent of the ocean has been identified as important and is not currently protected (**Figure 2**). It is encouraging that as the number of overlapping initiatives increased, so did the likelihood that the region was already within an MPA. This protection includes nearly all the areas identified as "high consensus". Only two-thirds of the current area covered by MPAs falls within regions that have been identified by the ten mapping initiatives; i.e., one-third of the current MPA area is not located within an identified area. Although we did not use maps from academic studies in our overlay, the areas we identified as "high consensus" – including the Galápagos Islands, the Caribbean Sea, Madagascar and the southern tip of Africa, the Mediterranean Sea, and the Coral Triangle – have also been identified by academic studies using diverse approaches and criteria (Selig et al., 2014; Klein et al., 2015; Martin et al., 2015; Jenkins and Van Houtan, 2016; Ramírez et al., 2017; Fischer et al., 2019).

Gaps in Current Protection

We focused our analysis on two major gaps in the current protection of identified marine areas: effectiveness of protection and representativeness of protection.

Effectiveness of Protection

No-take marine reserves are much more effective at achieving conservation goals than are other protected areas (Sala and Giakoumi, 2017; Zupan et al., 2018), yet we found that only 18% of the identified area was within MPAs reported as fully no-take. As a whole, only 2.2% of the ocean is strongly protected within no-take reserves (Marine Conservation Institute, 2019b), which is much lower than the IUCN-recommended level of at least 30% (IUCN, 2016). High-consensus areas were mostly protected on paper, but only 25% of the MPA area in these regions had a reported no-take status in WDPA. Though particularly evident in high-consensus areas, this was generally true for all identified areas; 57% of the total identified area did not have a no-take status and an additional 20% had a no-take status of "some."

Once an MPA is implemented, regardless of its no-take status, its conservation benefit is contingent on effective management and sustainable financing (Gill et al., 2017; Bohorquez et al., 2019). Factors like local buy-in, management capacity, and cost effectiveness are not part of the process of identifying important marine areas but are necessary considerations when choosing candidate sites for protection (Gilman et al., 2011; Schmitt, 2011). Improvements to and standardization of measures used to track the success of existing MPAs are also urgently needed (Fox et al., 2014; Woodcock et al., 2016; Giakoumi et al., 2018; Scianna et al., 2019). Though several tools exist for tracking effectiveness (e.g., How is your MPA doing? Pomeroy et al., 2004; Management Effectiveness Tracking Tool: Stolton and Dudley, 2016), evaluation of MPAs lags that of terrestrial protected areas (Global Database on Protected Area Management Effectiveness, 2019). Voluntary standards and certification schemes (e.g., IUCN Green List; Global Ocean Refuge System, 2019; IUCN, 2019), similar to Marine Stewardship Council for fisheries, may provide incentive to improve and track management in a greater number of MPAs.

The Galápagos province provides an example of a high-consensus area for which protection could be strengthened (**Figure 5**). Parts of the Galápagos were identified as important by seven of the ten initiatives we considered (EBSAs, PSSAs, WHS, AZE sites, IBAs, CI Biodiversity Hotspots, and WWF Priority Places). The province is also fully within an MPA, the Galápagos Marine Reserve, which includes subzones for conservation, tourism, and fishing (Castrejón and Charles, 2013). However, the no-take status of this reserve is unreported in WDPA. Ineffective enforcement and a lack of compliance within the reserve have been documented as limiting its ecological benefits (Castrejón and Charles, 2013; Buglass et al., 2018; Moity, 2018) and the reserve does not cover important habitat of several endangered species in the region (Ventura et al., 2019). Therefore, although the Galápagos province is fully covered by an MPA on paper, actual protection of this region appears insufficient.

Representativeness of Protection

All 99 of the ocean's provinces contained regions identified as important. However, over 50% of the protected identified area fell within just four provinces and more than half of all provinces had less than 10% of their identified area protected (**Figure 4**). These findings echo those of recent studies examining representativeness, which show that more than half of marine ecoregions fall short of the 10% protection target (Jantke et al., 2018), protection of the ocean's remaining wilderness is highly skewed toward a few biogeographic realms (Jones et al., 2018), and the ocean's geomorphic features and benthic habitat diversity are poorly represented in current MPAs (Fischer et al., 2019). We found similar results when considering representativeness across EEZs, with only a quarter of EEZs protecting at least 10% of their identified area.

Two regions we identified as moderate- to high-consensus with low rates of current protection were the Central Indo-Pacific and the Mediterranean Sea (**Figure 5**). Regions of the Central Indo-Pacific have been identified by both UN (EBSAs, PSSAs, WHS) and NGO (AZE Sites, CI Hotspots, IBAs) initiatives. Most of the MEOW and PPOW provinces in this region have an area-weighted average overlap of greater than one but protection levels of <5% (e.g., Eastern Coral Triangle, Indonesian Throughflow, Java Transitional). Additionally, several EEZs in the Central Indo-Pacific have <1% of their identified area protected (e.g., East Timorian, Malaysian, Papua New Guinea, Solomon Island EEZs).

The six Coral Triangle countries (Indonesia, Malaysia, Papua New Guinea, Philippines, Solomon Islands, and Timor-Leste), located in the Central Indo-Pacific, have developed their own regional initiative, the Coral Triangle MPA System. This initiative sets targets of protecting 10% of critical marine habitats in no-take reserves and 20% of critical marine habitats in some form of MPA by the year 2020 (White et al., 2014). Our analysis suggests that, according to WDPA's records, substantial progress must still be made to reach these targets (**Figure 5**). It is important to note that previous research has demonstrated discrepancies between various global and regional databases

tracking Coral Triangle MPAs (White et al., 2014). With the help of regional studies on spatial protection priorities (Asaad et al., 2018a,b), our results can provide guidance on selecting new candidates for protection.

In the Mediterranean, nearshore identified areas have relatively high rates of protection (14.4%) and an area-weighted average overlap of 3.8 (**Figure 5**). In contrast, deeper waters, which make up most of the Mediterranean and have a high average overlap (3.4), have very low rates of protection (3.6%). Among the many EEZs that fall within the Mediterranean, ten have <1% of their identified area protected. Shortcomings in the Mediterranean network of MPAs have been previously noted, including a low coverage of areas that prohibit all extractive activities (0.1% as of 2016), the predominance of small MPAs (<50 km^2), and a lack of effective management (Bastari et al., 2016; Rodríguez-Rodríguez et al., 2016a). However, there is also recognition that large portions of the Mediterranean (~61% of the total area) are closed to bottom trawling and, while this protection may not currently count toward global conservation targets, it is likely to have ecological benefits (Rodríguez-Rodríguez et al., 2016b).

There is growing interest in management of ABNJ and negotiations are underway to develop legally binding conservation measures for these areas under the United Nations Convention on the Law of the Sea (UNCLOS) (UN General Assembly Resolution 72/249; Heffernan, 2018; Wright et al., 2018; Tiller et al., 2019). A focus of these negotiations is area-based management. To date, ABNJ MPAs have only been possible through sectoral organization designation, including designations by CCAMLR or the Convention for the Protection of the Marine Environment of the North-East Atlantic (the OSPAR Convention) (Smith and Jabour, 2017; Tiller et al., 2019). A unified selection process has been noted as a need for the establishment of ABNJ MPAs (Smith and Jabour, 2017), and our study highlights over 76 million km^2 of unprotected ABNJ that have already been identified as important and that can act as a starting point for future deliberations.

Study Caveats

The "best available science" is often imperfect science. Geographic and taxonomic data gaps (Gilman et al., 2011) and other limitations of the initiatives we collated influenced our

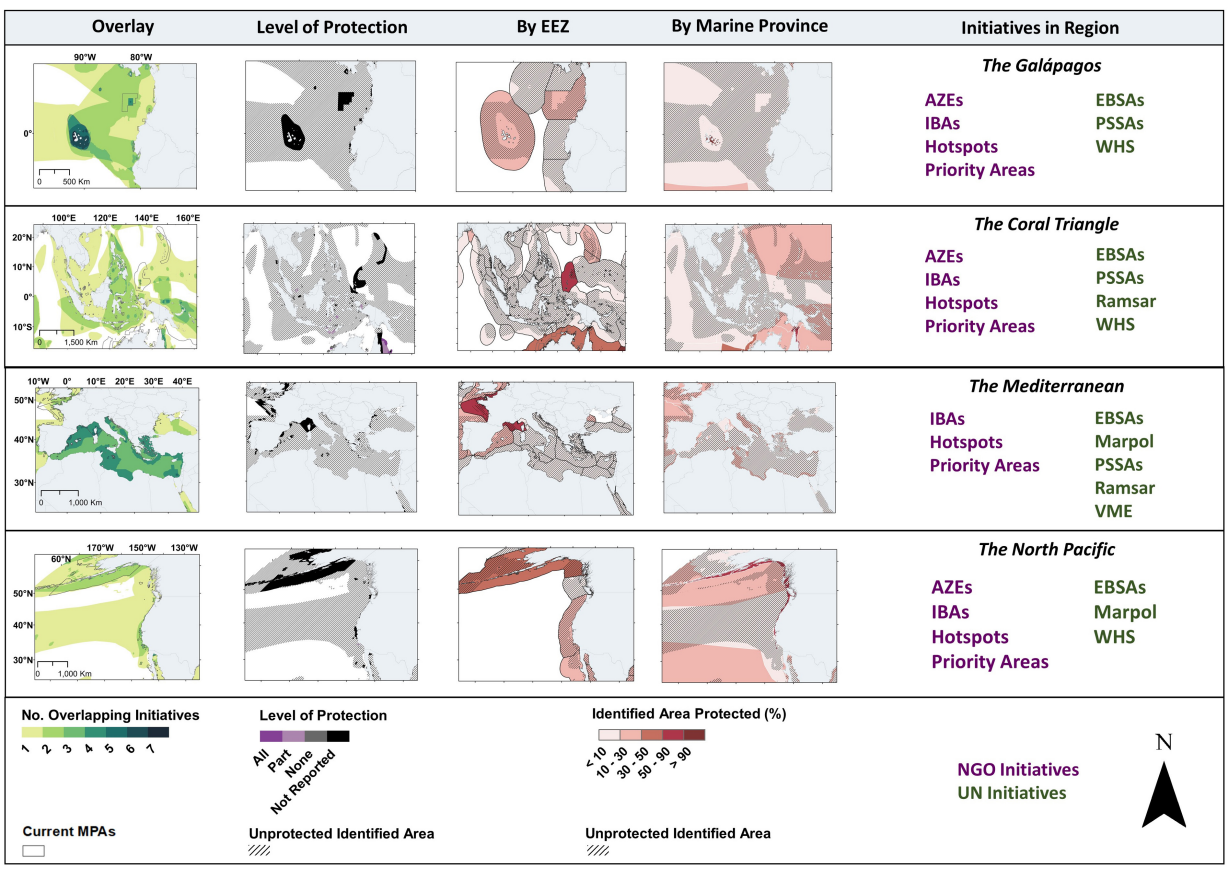


FIGURE 5 | Examples of important marine areas with opportunities for stronger, more representative, and/or more widespread protection. Areas were identified as important by six United Nations and four non-governmental organization initiatives that vary in purpose and scope. For each region, information is provided on the number of overlapping initiatives, on the strength of current protection, on identified areas within marine provinces (MEOWs, Marine Ecoregions of the World; Spalding et al., 2007; PPOWs, Pelagic Provinces of the World: Spalding et al., 2012), and on identified areas within exclusive economic zones (EEZs, Exclusive Economic Zones; Flanders Marine Institute, 2018). Countries are displayed using data provided by Natural Earth (Admin 0 Countries 4.1.0; Natural Earth, 2018a).

results. For example, current prioritization initiatives might be missing important regions of the ocean by focusing primarily on species-level diversity, ignoring biodiversity "coldspots" and genetic diversity as a result (Marchese, 2015). We are constantly improving our knowledge of marine ecosystems. Global initiatives to identify important marine regions should therefore be an ongoing process that take these new data and insights into account.

Here, we use the EBSA identification process to illustrate some of these caveats, as this process has been well-documented (Dunn et al., 2014; Bax et al., 2016; Johnson et al., 2018). EBSA workshops rely on the input of a wide range of stakeholders who undergo a pre-workshop training process and collate all relevant global and regional spatial data (Bax et al., 2016). As of early 2018, 291 EBSA workshops had been held around the world (Johnson et al., 2018). The resulting EBSA map is one of the most comprehensive initiatives we considered in our study.

Several regions were excluded in the EBSA process at the request of State parties (e.g., United States EEZ) or because national governments were not represented at workshops (e.g., the Pitcairn Islands) (Johnson et al., 2018). Relevant spatial data were notably lacking in some of the considered regions (e.g., Southern Indian Ocean) or habitats (e.g., pelagic zones) (Johnson et al., 2018). There were several taxonomic groups underrepresented in this process, including corals, elasmobranchs, structure-forming plant habitats, and large cold-blooded reptiles (Johnson et al., 2018). As with other prioritization initiatives, there were also gaps in the types of habitats considered during the identification process (Briscoe et al., 2016). The first map of global mesopelagic biogeographic zones was published less than 2 years ago (Sutton et al., 2017) and will be considered in future EBSA workshops (Johnson et al., 2018). Lastly, IBAs were part of the EBSA decision-making process, leading to some redundancy. IBAs and EBSAs overlapped in 18% of the total area we found to be of moderate or high consensus.

Prioritization initiatives are based on historical abundance information, which is generally collected over a short time period (Gilman et al., 2011). Biodiversity data collected over limited time scales can mislead prioritization in systems that are highly variable over space and time (Piacenza et al., 2015) and do not account for the future state of the ocean. The ocean is changing rapidly due to climate change and other anthropogenic impacts (Halpern et al., 2015; Jones et al., 2018). Although MPAs can increase ecosystem resilience in the face of climate change (Roberts et al., 2017; Sala and Giakoumi, 2017), marine habitats and species are likely to migrate in coming decades as a result of changing ocean conditions (e.g., thermal stress: Fredston-Hermann et al., 2018 but see Davies et al., 2017). Under business-as-usual emissions, sea surface temperature and oxygen concentrations will fluctuate beyond their natural limits by 2050 in 42% of MPAs (Bruno et al., 2018). Climate change may unevenly affect the areas identified by initiatives depending on the criteria used to identify them. For example, in terrestrial systems, criteria related to rarity correlate with how robust "hotspots" are to climate change; this correlation likely exists because regions with stable environments foster specialists, so are more likely to fit these selection criteria (Iwamura et al., 2013).

Lastly, the accuracy of our gap analysis is dependent on timely and accurate reporting to the WDPA by member states. There are often lags between the designation of protected areas and their inclusion in the WDPA, although the length of these lags is decreasing (UNEP-WCMC and IUCN, 2018b). There have been documented issues with under-estimation (Visconti et al., 2013) and over-estimation of some MPAs in the past (Smallhorn-West and Govan, 2018). The process for country reporting to the WDPA can be improved in various ways, but the dataset itself is updated monthly, well managed, and continually being improved. For example, Smallhorn-West and Govan (2018) raised concerns about the overreporting of Tonga's MPA coverage. The country's MPA area represented in the WDPA subsequently decreased from over 10,000 km^2 in January 2018 to 35 km^2 in January 2019 (Smallhorn-West and Govan, 2018).

We did not consider proposed MPAs, i.e., sites for which policymakers have announced their intent to designate an MPA, in our gap analysis. MPA pledges currently identified by the WDPA (as of March 2019) account for 7.4% of the global ocean, in addition to the 7.6% of the ocean that exists in MPAs that are already implemented. Spatial data do not yet exist for most proposed MPAs. However, the Marine Conservation Institute (2019a) provides a general map of regions with ongoing campaigns seeking additional protection. Several of these campaigns fall in regions revealed to be of moderate- or high-consensus in our analysis, including regions in the Mediterranean, in the Coral Triangle, in the Caribbean, surrounding Madagascar, and along the coasts of Chile and Argentina (Marine Conservation Institute, 2019a).

Next Steps

The initiatives we considered varied greatly in purpose and scope, but all relied on specific criteria and expert review processes to identify important areas (e.g., Dunn et al., 2014; Diz et al., 2018). Even areas identified by one initiative (40% of the ocean) could provide an opportunity for regional efforts to identify new MPAs. Additionally, 14.5% of the ocean (52.5 million km^2) has consensus among two to four initiatives, and most of this area remains unprotected (**Figure 2**). To identify new MPA sites, however, our overlay map must first be refined for smaller geographic scales using data collected at higher spatial resolution.

Much of the area identified lies within contiguous stretches of $>$100,000 km^2; it is unrealistic to propose that all this area be protected. Because these areas are large, however, they provide flexibility in adding to a global network of small and large MPAs to meet the following criteria: representativeness, replication, ecological connectivity, size, and refugia from climate change-related effects (Gilman et al., 2011; Green et al., 2014). Furthermore, nearly every EEZ and biogeographic province has area identified as important but not currently protected. Regions we identified as moderate consensus, i.e., areas with agreement among two to four mapping initiatives, are of particular interest. In some cases, these regions lie in areas dominated by one EEZ (e.g., the Madagascan EEZ). In many cases they occur in areas

consisting of several EEZs, which could provide opportunities for regional collaboration toward ocean conservation.

In some of the large identified areas, large MPAs would be difficult to enforce or have excessively negative socioeconomic impacts, so networks of small MPAs would be the most ecologically beneficial (e.g., in the Coral Triangle: Walton et al., 2014). Our overlay map would need to be combined with detailed information on the movement patterns of species in the region to guide MPA size, shape, location, and spacing (e.g., Green et al., 2014; Munguia-Vega et al., 2018). For example, movement distances across 210 species of coral reef fish range from less than 0.5 km to greater than 1,000 km (Green et al., 2015).

Large MPAs, which are not without criticism (Leenhardt et al., 2013), can complement networks of small MPAs because they provide unique benefits such as buffering against uncertainty, especially in the case of climate change-induced shifts in species' ranges (Davies et al., 2017; O'Leary et al., 2018). Large MPAs are likely to be most appropriate in pelagic habitats, where they can benefit highly migratory species (Davies et al., 2018) and provide protection of oft-ignored oceanographic features like fronts and eddies (Briscoe et al., 2016). For example, Harrison et al. (2018) found that 18 migratory species collectively visited EEZs of 37 separate countries within the Pacific Ocean and that nearly half of all individuals tracked spent time in ABNJ. Two of the largest identified areas we found were located in the Pacific Ocean (**Figure 5**) and, when combined with other data (e.g., Tagging of Pelagic Predators project[1]), could be used to identify new candidates for large-scale MPAs.

In the case of either MPA networks or large-scale MPAs, analyses such as ours cannot account for regional complexities associated with MPA designation and management, as alluded to in our examples of the Galápagos, Coral Triangle, and Mediterranean. On-the-ground realities of how marine areas are used will be important when considering where to designate new MPAs and how to protect areas of high ecological value. However, the global MPA network is rapidly changing, and the map resulting from our analysis can act as an immediate starting point for systematic conservation planning (Margules and Pressey, 2000; Ban et al., 2009; Watts et al., 2017). For example, "From the Vision to the Ground" is a guide produced by WWF (Loucks et al., 2004) that outlines the steps that should be taken to scale global conservation initiatives down to the landscape level. This process includes mapping social landscapes and working with a diverse set of stakeholders to develop conservation plans (Loucks et al., 2004).

CONCLUSION

It is imperative that protected areas be based on the best science available, and this sentiment is contained in ocean protection targets. Over the past two decades, various UN and NGO groups have put substantial effort into identifying areas of the ocean that warrant special consideration. As nations and international bodies seek to achieve numerical protection targets, they will look to the resulting maps for guidance. Our study was conceived after discussions with policy makers who requested direction on how to reconcile multiple global maps. While further information will need to be incorporated at national and regional scales to identify MPA sites and networks, we hope that the mapping overlay presented here will provide valuable insights into where there are already areas of consensus and opportunity.

[1]http://gtopp.org/

DATA AVAILABILITY STATEMENT

The spatial data resulting from this study are available on request to the corresponding author. The spatial data used as inputs to the study can be found online or by contacting the organization that created the data. Links to the input data sources can be found in **Supplementary Appendix S3: Supplementary Table S1**.

AUTHOR CONTRIBUTIONS

EP conceived of the study. NG conducted the data analysis. EP, CS, and JD provided guidance and feedback on data analysis. NG, EP, CS, and JD wrote and revised the manuscript. EP and CS obtained funding for the work.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00650/full#supplementary-material

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To Ban or Not to Ban? Reviewing an Ongoing Dilemma on Sea Turtle Egg Trade in Terengganu, Malaysia

Jarina Mohd Jani[1,2†], Muhammad Allim Jamalludin[1] and Seh Ling Long[3]*

[1] Faculty of Science and Marine Environment, Universiti Malaysia Terengganu, Kuala Terengganu, Malaysia, [2] Institute of Tropical Biodiversity and Sustainable Development, Universiti Malaysia Terengganu, Kuala Terengganu, Malaysia, [3] Institute of Oceanography and Environment, University of Malaysia Terengganu, Kuala Terengganu, Malaysia

Legal trade in sea turtles and their eggs remains a reality in many countries where conservation of this marine endangered species does exist. This duality is a conflict to some who appeal for a total trade ban, which may have implications on local livelihoods. Using the Sustainable Livelihoods Approach (SLA), this paper considers the dilemma by empirically examining the context of the state's legislation, particularly the Terengganu Turtle Enactment (TTE) and its "haves" – what is legally provided on paper, the available capitals in hand, and the actual implementation carried out in practice – to elucidate whether a trade ban is indeed the immediate solution for improving sea turtle conservation in the state of Terengganu, Malaysia, which is an important rookery in Southeast Asia. Findings based on data collected through extensive archival research and in-depth interviews with officers of the state who manage sea turtles as well as those whose past and current livelihoods depend on the trade indicate that (1) sea turtles are a transformative natural capital that the Terengganu legislation supports on paper via a pro-conservation concession system; (2) in putting paper into practice, those involved in its implementation have strategically mobilized available resources to achieve a balanced outcome between conservation and livelihood; and (3) institutional absenteeism, financial handicap, and ambiguous legal protection status of nesting beaches are issues that need addressing to fulfill the true potential of this legislation. However, a better conservation outcome is administratively possible via a full conservation–concession system that lists all beaches under the TTE to ensure the sustainability of sea turtles and local livelihoods, in preparation for the ultimate long-term goal: total conservation via a trade ban without which conservation of the species remains precarious not only at state but also at national and regional levels.

Keywords: sea turtle conservation, turtle egg concession, Terengganu, trade ban, Southeast Asia, sustainable livelihoods approach

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****Correspondence:***
Jarina Mohd Jani
jarina@umt.edu.my

†ORCID:
Jarina Mohd Jani
orcid.org/0000-0002-2126-7025

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INTRODUCTION

Sea turtles are an iconic megafauna that are now protected from international trade in 178 signatory countries of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) (Humber et al., 2014). The seven known species of sea turtles, leatherback turtle (*Dermochelys coriacea*), loggerhead turtle (*Caretta caretta*), flatback turtle (*Natator depressus*), olive ridley turtle (*Lepidochelys olivacea*), Kemp's ridley turtle (*Lepidochelys kempii*), green turtle

(*Chelonia mydas*), and hawksbill turtle (*Eretmochelys imbricata*), are, however, within the national boundaries of the countries where they occur, subject to different levels of protection, often depending on the local sea turtle–human interaction context (Frazier, 2003). In many important sea turtle rookeries in the world, traditional consumption of either their eggs, meat, or both are legally protected (Campbell, 2010; Garland and Carthy, 2010; Grayson et al., 2010), while their shells continue to be a popular commodity in East Asian markets (Lam et al., 2011). Furthermore, there is also an increasing demand for allowing cultural use in countries where total protection is provided (Rudrud, 2010). Along the West African coast where they are consumed not only for food but also as an important ingredient for traditional healing, Fretey et al. (2007) suggested that exemption be given for cultural uses. Due to the significance of their consumptive use in some indigenous communities, it is defended as a cultural right that makes legal prohibitions problematic (Barrios-Garrido et al., 2017). These various instances of legal consumptive use of sea turtles and calls made in their favor often do not sit well among conservationists, where they are denounced as pejorative, not complimentary elements to sea turtle sustainability (Campbell, 2002; WWF-Malaysia, 2015). In the case of Malaysia, an important sea turtle rookery in Southeast Asia where four species occur, a conservation versus concession debate rages on due to the legal trade of sea turtle eggs that is claimed to impede the sustainability of the megafauna in the country (WWF-Malaysia, 2012).

In Malaysia, sea turtle eggs have always been the only consumed animal part (Hendrickson and Alfred, 1961; Chan and Liew, 1996) that have been locally traded and regulated by local laws before the colonial era (Hendrickson, 1958). Since Malaysia's independence, every state in Peninsular Malaysia may exercise their right to set up rules regarding sea turtles and their eggs with the powers conferred to them by the Fisheries Act 1963 and its amendment in 1985 as the animal is constitutionally considered a natural resource that the federated states can decide how best to manage (see Gregory and Sharma, 1997 for the sea turtle legislation scenario within the state-federal division of law in Malaysia). As a result, the sea turtle egg consumption and trade is either totally banned (in Sabah and Sarawak), not at all regulated (in Perlis and Selangor where turtle nesting is insignificant), or regulated through trade concessions (in the remaining nine federated states). Among those in the third category, Terengganu makes the most interesting case study not only due to its place in sea turtle conservation history as the most famous leatherback turtle rookery in the world (Hendrickson and Winterflood, 1961; Chan et al., 1988; Chua and Furtado, 1988) that met a tragic end, but also due to its current importance as one of the major nesting sites for green turtle in the Southeast Asian region (Chan, 2006). Here, conservation efforts on protected nesting areas in Terengganu continue mostly for green turtles through beach monitoring and hatcheries (Chan, 2013; Abd Mutalib et al., 2015). But the trade of sea turtle eggs sourced from legally tendered nesting beaches remains legal and is said to pose a potential threat to the sea turtle population. As much as 422,000 sea turtle eggs were reportedly sold in local markets in 2007 alone (TRAFFIC Southeast Asia, 2009). For this reason, there is much pressure from the sea turtle conservation front to push for stricter legal measures in addition to those that have been put in place by the Terengganu Turtle Enactment (TTE) in 1951 (The Star, 2005; WWF-Malaysia, 2010). The ban on the sale and consumption of sea turtle eggs should, according to scientists and conservationists, be extended to the other three remaining species (Aikanathan and Mortimer, 1990; Chan and Liew, 1996; Chan, 2006; TRAFFIC Southeast Asia, 2009; WWF-Malaysia, 2010), while the tender system must be stopped to avoid the same tragedy with the leatherback turtles (Ibrahim and Sharma, 2006). These anti-concession calls stressed that the legislation of turtle egg exploitation via a tender system jeopardizes the conservation efforts, mainly due to the lucrativeness of the trade (WWF-Malaysia, 2012). However, the state has a different view regarding the matter, i.e., such radical action could negatively affect the culture and the livelihood of the people in Terengganu. The state's Chair of Agriculture and Regional Development Council once stated that such a ban would not solve the problem, but could in fact raise market prices and encourage poaching (Kent, 2006). Another representative of the state had confirmed that while the sale or consumption of sea turtle eggs is not encouraged, the state did not plan to ban these practices that are part of Terengganu's tradition (WWF-Malaysia, 2012). Indeed, the sea turtle egg trade not only concerns the turtle's survival but also local community livelihoods. During a media interview, a local trader strongly opposed the idea of banning the trade because it is a source of livelihood, which he did not believe to be a threat to sea turtles' survival (Kuppusamy, 2012). All these pro-concession perspectives demonstrate that the complexity of the issues revolving around human–sea turtle interactions in the sea turtle egg trade involves not only the ecological principles but also the cultural, social, economic, and political factors, which have led to an endless debate without any solutions so far in sight. But a better understanding of what actually is already in place from the legal and operational perspectives of sea turtle conservation as well as their reasons why – gained using an approach that appreciates both the value of sea turtle conservation and local livelihoods – will help in finding a constructive next step.

In this paper, the request for "more law" in sea turtle management in Terengganu is considered by examining its policy dimensions based on the Sustainable Livelihoods Approach (SLA). This is primarily because in SLA, the focus has intuitively always been on the "haves" rather than the "have nots" (Bernstein et al., 1992), which the authors consider to be a more practical approach to the debate than the current focus on what is lacking in the state's sea turtle egg trade policy. According to DFID (1999), "a livelihood comprises the capabilities, assets (including both material and social resources) and activities required for a means of living. A livelihood is sustainable when it can cope with and recover from stresses and shocks and maintain or enhance its capabilities and assets both now and in the future, while not undermining the natural resource base" (Section 1.1; see also Scoones, 1998, and Carney, 1998, p. 4). The focus of this paper is on the access component, which consists of elements that mediate the social–institutional and political processes that essentially link resources to strategies that are deployed to achieve the desired livelihood outcomes (Scoones, 2015). By analyzing the

legal framework of the state's sea turtle management, as well as its actual implementation by state actors, this paper aims to establish what is already in place, on paper and in practice. In presenting explanations of why is it so, this approach allows the authors to review the elements of assets that the various stakeholders have in strategically responding to the Terengganu sea turtle egg trade policy. It then makes it empirically comfortable to consider its efficiency before recommending the way forward, be it more law or not. By doing so, the paper provides an objective insight on the concession versus conservation debate with hope to contribute toward constructive actions to ensure a sustainable future for both local livelihoods and sea turtle populations in Terengganu as well as in other countries where the dialogue on trade bans is taking place.

For answers, the paper engages with relevant documents on the legality of the sea turtle egg trade, and its operation on the ground, and with actors from both sides of the debate, i.e., state officers in charge of sea turtle conservation or trade, past and current trade concessionaires, those involved in harvesting the eggs, and traders who sell them in the market. It begins with a brief description of the study's SLA-based theoretical framework and continues by explaining the research methods. Then, we present and discuss the results from reviewing the existing legal and policy framework for sea turtle management, as well as their actual implementation for providing recommendations toward resolving the trade ban debate.

THEORETICAL FRAMEWORK: SLA-BASED SEA TURTLE POLICY ANALYSIS

Sustainable livelihoods approach has now developed as a concept and framework (Ashley, 2000; Krantz, 2001) as well as a set of "principles for action" (Toner, 2003, p. 772), and has been applied in designing and reviewing projects, programs, or sectors (Farrington et al., 2004). The strength of livelihoods as a framework lies in its core concepts: firstly to give focus on people, secondly to be holistic in application, and finally to provide the links between the micro, i.e., lived realities of local resource users; and macro, i.e., state level policies and structures; dimensions that are indispensable in resource management (Farrington et al., 2004). According to Murray (2001), its dynamism and the analysis of strengths first rather than needs has provided the framework with many advantages, such as (i) emphasis to understand livelihood diversification in adapting to change in both the present and the past contexts; (ii) advocacy for stretching analysis between different (micro to macro) levels; (iii) acknowledgement of the transboundary nature of different sectors; and finally (iv) recognition given to examining the relationship between the various livelihood strategies within a studied unit and its social interaction with other units of study. Although more widely applied in terrestrial settings, its adoption as the conceptual framework for marine research has allowed for the consideration of both coastal livelihood needs and marine ecological concerns, which are indispensable in livelihoods that are subject to natural resource cycles and fluctuation such as the fisheries sector (Allison and Horemans, 2006). Recently, SLA has also been applied to sea turtle field research to understand the link between community livelihoods, community well-being, and sea turtle conservation where a sea turtle conservation program operates (Montoya and Drews, 2006). The link between "asset-access-strategies-outcome" in SLA analysis provides a practical research *modus operandi* in unfolding the often intricate reality of marine endangered species protection and people's livelihood. A broader analytical approach, the use of SLA framework for analysis permits the authors to consider the Terengganu legal institution "as the network of rules and patterns of behavior that condition local resource use and decision making" (King, 2011, p. 299) instead of a mere legal structure and procedures.

This paper is drawn from a study on the human ecology of sea turtle conservation in Terengganu that used this approach to facilitate the process of elucidating the inter-relationships between the livelihood assets, access, strategies, and expectations of resource managers and users. More explicitly, the study examined the various capitals that sea turtle resource managers on one hand and users on the other have, the mediating factors that govern how they apply these capitals into strategic activities, either toward conservation or concession, in order to achieve the outcome that they expect. But this paper focuses primarily on the access component, i.e., the policy on the Terengganu sea turtle egg trade. It looks at the policy framework as an active arena for sea turtle conservation and concession-based livelihood making where there is no tangible "unit of study" such as households or individuals. Instead, this analysis considers the legal "on paper" TTE provisions as the access elements for all those involved in the Terengganu sea turtle egg concession and/or conservation while what is put in practice are "strategies" deployed by these various stakeholders, particularly the Department of Fisheries (DoF), in realizing the outcomes they hope for, i.e., livelihood and sea turtle sustainability (**Figure 1**). Based on the understandings gained on the available resources, i.e., assets that these actors have, the paper discusses the reasons for their choice. This contributes toward a more holistic evaluation of the current performance of the Terengganu sea turtle management efforts. It therefore begins by analyzing the core of the legal framework, i.e., the provisions in the TTE in its various versions that is the basis for managing sea turtle egg trade and their actual implementation, i.e., the responses of the institutions, organizations, and entities that constitute the active component of the policy dimension in order to later infer on the performance of this legislation.

Qualitative Data Collection

This paper is based on data collected since November 2014 to May 2018 through various materials and methods. It draws firstly on library and archival materials dated between 1951 and 2018, covering primarily three legal documents, i.e., the TTE in 1951 and its amendments in 1987 and 1989, as well as the tender notice and contract, reports, and minutes of meetings compiled for the Terengganu Turtle Sanctuary Advisory Council (TSAC) from 1988 until 2014, as well as nesting and hatchery management reports and related official documents on the concession or conservation activities (refer to **Supplementary Table S1** for the full list). These documents were sourced mainly

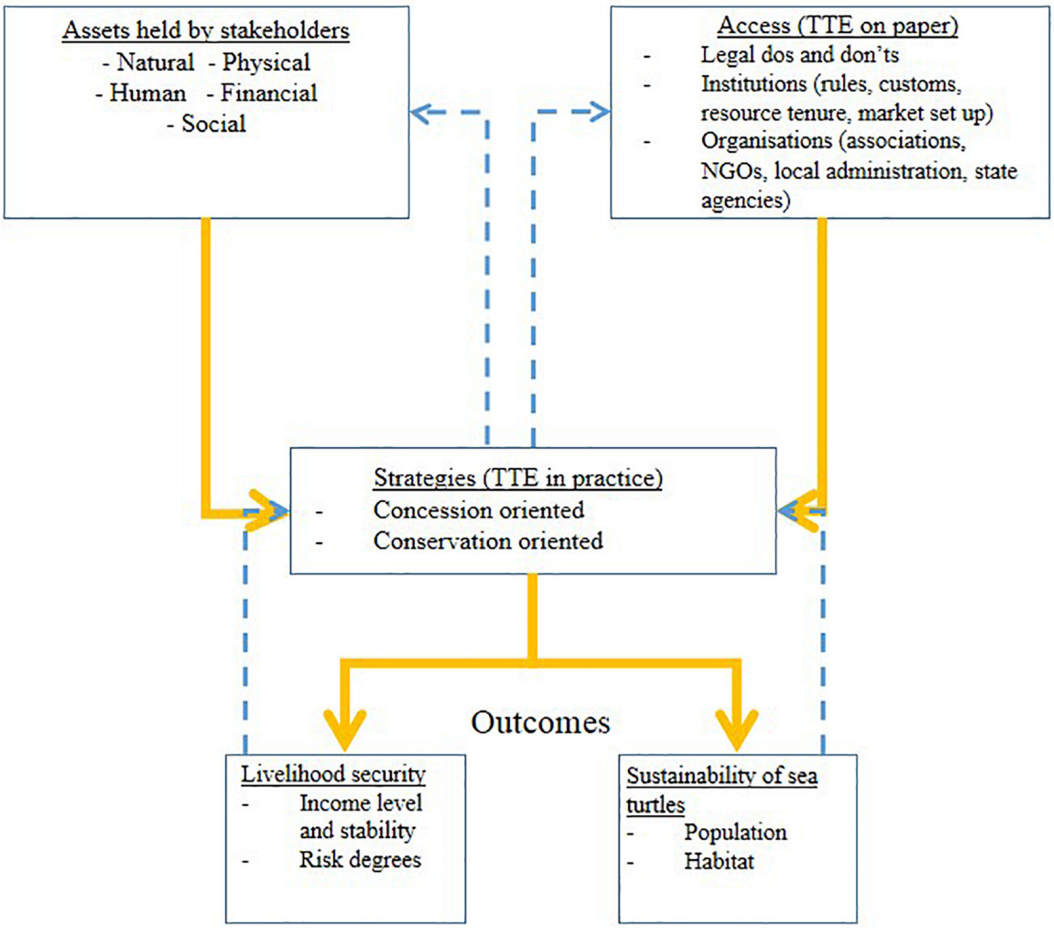


FIGURE 1 | Sustainable livelihoods approach-based theoretical framework for evaluating the Terengganu turtle egg trade policy. Yellow arrows refer to influence while dotted blue arrows refer to response.

from government agencies and NGOs. Literature relating to the legal framework that structures the sea turtle egg concession and conservation in Terengganu since TTE was first introduced was also reviewed.

Secondly, to understand the implementation of sea turtle concession and conservation in reality, the study collected data in the Terengganu mainland and marine park islands by carrying out in-depth qualitative interviews with key informants, i.e., individuals who are directly involved in sea turtle concession and conservation to capture actions and practices of those affected by the legislation. In the mainland where the tender system still exists, the focus was on the practices related to the various activities – from harvesting to retail sale – at licensed beaches and markets as well as reserve and hatchery management. The interviews were conducted from November 2014 until November 2015 with nine sea turtle egg traders based in the main market (Pasar Payang) where most sea turtle eggs are sold, as well as 13 out of 18 registered license holders, of which 6 depended fully on employed egg collectors at their licensed beaches. Three of these employed egg collectors were also interviewed as well as three state government officers. Meanwhile, between October 2016 until April 2018, 70 households were surveyed in the state's main marine park islands of Redang and Perhentian – location of important sea turtle rookeries where the tender system used to exist until 2006 – particularly on the local communities' perception on the impact of the sea turtle nest conservation in these marine protected areas. All respondents were recruited opportunistically, sometimes through snowball sampling due to the sensitivity of the subject. Also included were field notes taken during participant observations in various locations related to the study, from market places to conferences. The data collected were then categorized and later analyzed according to the SLA themes framed by the research framework, i.e., the asset-access-strategies-outcomes linkages that form the complex policy dimensions between what is legally provided by the TTE and the actual practice (see **Figure 1**). Specifically, the analysis first concentrates on the access component of the SLA framework, explaining provisions in the TTE for sea turtle concession and conservation. It then reviews the governing and/or institutional structures that are in place and the strategies that are deployed on the ground. Finally, a comparison of what is legally provided

on paper versus what is actually implemented on the ground is made to identify gaps hampering full protection for sea turtle eggs in Terengganu.

RESULTS

The results are presented in three subsections. The first subsection shows how the laws have evolved from a purely trade-oriented legislation to one that increasingly became supportive of conservation. The paper then explains in the following subsection that a transition from consumptive to non-consumptive use of sea turtles has resulted as the stakeholders' strategic adaptation to this legal evolution. Finally, the identified gaps between "paper and practice" – due to the absence of certain governing/institutional structures as well as the lack of certain assets that force those involved to make do with what they have – are presented in the third subsection. Through these findings, the outcome that could be expected from the current management of the sea turtle egg trade is then discussed.

TTE on Paper: A Trade Oriented Legislation That Has Evolved in Support of Conservation

Although there was a provision in the first TTE to create notified areas where no eggs could be taken (no take zone) under Section 8, this state legislation was promulgated in 1951 principally as a legal mechanism to regulate the trade of sea turtle egg. For this reason, the early provision of this law was to enable the establishment of a state governing body of the trade mainly through a licensing mechanism of turtle nesting beaches. It was then a purely trade-oriented legislation that did not cater to any matter beyond managing the allocation of "listed beaches" through concessions earned via commercial tender or exclusive rights. It was, however, considerably amended in 1987 and once again in 1989 to enable the inclusion of a new element of sea turtle management in the state, i.e., turtle sanctuary beaches. Here, eggs nested within the area are fully incubated and the activities allowed within the sanctuary are regulated. Since then, the trade concession of sea turtle eggs, termed concession henceforth in this paper, began to coexist with conservation efforts to protect the same resource. The chronological progress of this transformation is summarized in **Figure 2** while **Table 1** presents the different categories of nesting beaches that now exist in the state.

Terengganu turtle enactment is in fact a sophisticated tool for sea turtle conservation for two reasons. Firstly, it has various provisions that, although were originally meant for managing sea turtle egg concession, could also be used for conservation, as summarized in **Supplementary Table S2**. In general, the law ensures that not everyone has access to egg collection, except in "non-listed areas," i.e., beaches that are neither legally designated for concession nor conservation. Section 7 states that only those with a license, their employees, or the ones given explicit permission to do so are allowed to collect sea turtle eggs at listed beaches for concession or conservation purposes, hence limiting

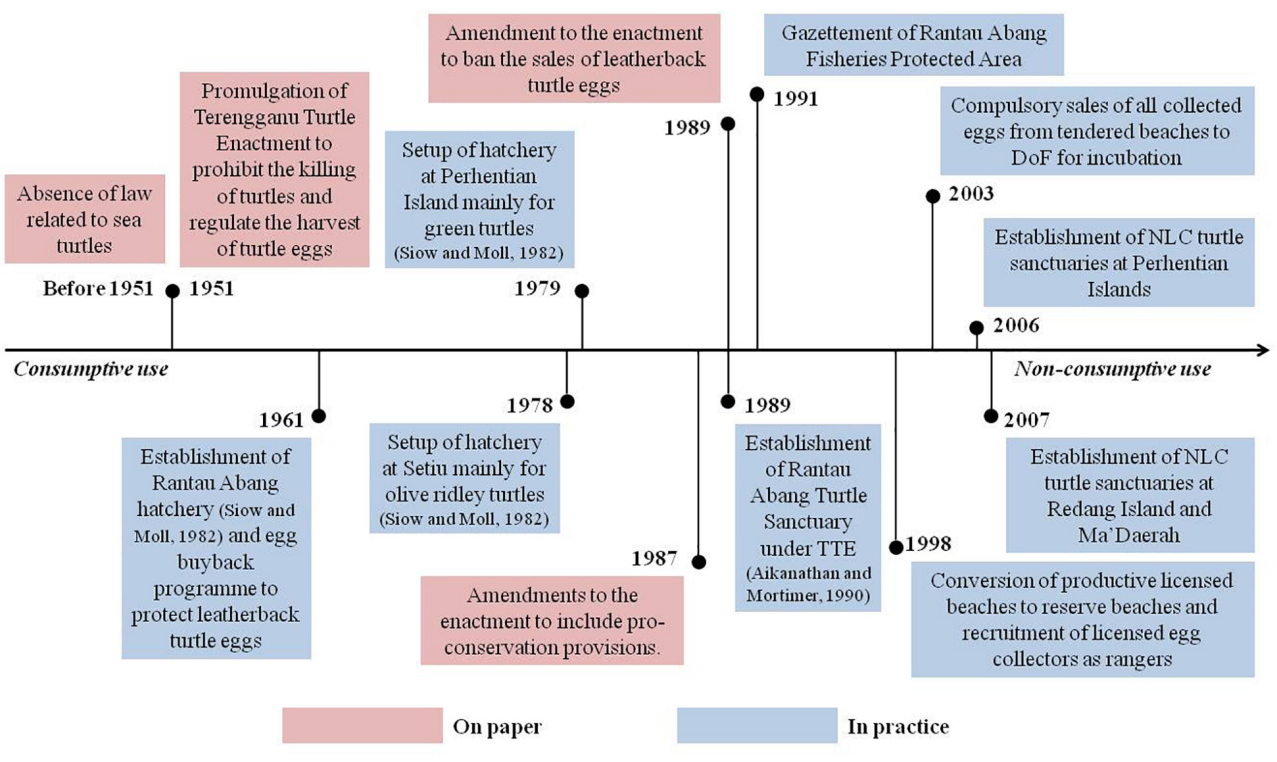


FIGURE 2 | Chronological progress of TTE transformation in practice.

TABLE 1 | Categories of nesting beaches that exist in Terengganu.

Category	Definition	Status
TTE sanctuary	Nesting beaches that have been gazetted for conservation as Turtle Sanctuary under Section 3A of TTE.	LISTED
NLC sanctuary	Nesting beaches that have been gazetted for conservation as Turtle Sanctuary under the National Land Code.	LISTED
Notified areas	Areas designated by the King as no take zones (i.e., for conservation such as sanctuaries and reserves) under Section 8 of TTE.	LISTED
Licensed	Nesting beaches that have been listed in the tender list for egg collection licensing purposes. Although originally for concession (i.e., trade), eggs from these beaches could be collected for conservation as well.	LISTED
Reserve	Nesting beaches that were formerly listed for tender but have been delisted and put under DoF's management for conservation.	LISTED
No status	Other beaches in Terengganu that have no official status under TTE or NLC. Eggs found on these beaches are not protected and can be collected by anyone.	NON-LISTED

access to this resource. Any other person who is suspected to take the eggs in these listed beaches could be taken into custody for investigation and prosecution not only by the Licensing Officer or the Authorized Officer but also a member of the police force. Furthermore, as mentioned earlier, Section 8 makes it possible for the creation of no-take zones in the state for *in situ* incubation, while Sections 9 and 10 could give license or exclusive right to collect egg for conservation purposes. Other provisions that also benefit conservation are those related to offenses and penalties, i.e., Sections 5, 6, 7, 12, 14, and 15.

Secondly, it is found that while the enactment has not made any additional amendments to favor sea turtle egg trade, it has done significantly so for conservation since 1987. In response to the drastic decline of the leatherback turtle population recorded in the 1980s, the 1951 version was extensively adjusted mainly to accommodate a major conservation move, i.e., the establishment of Rantau Abang Turtle Sanctuary under Section 3A in 1987 to make protecting nesting beaches for conservation possible. They resulted in the following three major improvements:

(i) Inclusive management – the addition of Sections 3B, 3C, 3D, and 3E enabled a more inclusive governance model in the form of the Turtle State Advisory Council (TSAC) whose membership was extended not only to relevant state agents but also the academia and conservation NGOs. Also, the establishment of a local management body via the Turtle Steering Committees at district level was made possible.

(ii) Enhanced enforcement manpower – new provisions under Section 4 created the appointment of various officers and their jurisdiction in enforcement, in particular the Authorized Officer. The section further facilitated the enforcement by the Police where all ranks of the corps could enforce this legislation – not only inspectors and above as stated in the previous version of the law.

(iii) Strengthened protection for sea turtles – increased penalties in Section 5 for killing and taking turtles as well as the prohibition of carrying turtle-based tourism activities on any nesting beaches without a license (Section 12A). Finally, in 1989, a total ban on leatherback turtle egg consumption under Section 7A was established – a historical legislation, being the first of its kind in Malaysia and remains a unique legislation because, to date, no other legal ban has been introduced in any of the federated states of Peninsular Malaysia.

Indeed, our findings suggest that in four decades, the TTE has evolved from a mere trade-oriented legislation to one that provides for the implementation of protective measures for the sea turtle population in the state. We found that the evolution of the TTE legislation is progressive due to its receptiveness to scientific reasoning without losing its sensitivity to local traditions, i.e., sea turtle egg consumption. Indeed, based on the reports that were reviewed, the change in the law and in practice appears to have been influenced by the works of sea turtle scientists and conservationists working in the state, who pushed for a top-down move by demonstrating the value of conserving the species. Studies in the 1980s (Siow and Moll, 1982; Mortimer, 1988) contributed toward including provisions for conservation in the enactment that was fully supported from the legal and institutional aspects to ensure the balanced coexistence between concession and conservation of sea turtle eggs. Meanwhile, those in the subsequent decade (see Chan et al., 1985; Chan et al., 1988; Mortimer, 1989, 1990; Aikanathan and Mortimer, 1990; Mortimer and Daud, 1991) successfully pushed the conservation agenda further by influencing the state to use the administrative means provided by the law to move toward a unique pro-conservation concession system through the introduction of mandatory sale of all collected eggs to the state's DoF. Their engagement with decision-makers at the right moment, such as providing the hard scientific proof of the leatherback turtle's drastic demise, has helped to improve policy making.

TTE in Practice: Transforming Sea Turtles Eggs From a Consumptive to a Non-consumptive Resource

What has been the impact of this legislation on stakeholders, i.e., agents of the state responsible for managing the concession and conservation of sea turtle eggs, conservation organizations, as well as local communities that depend on the sea turtle eggs for their livelihoods? The chronology of important events that has taken place since its promulgation does suggest that it has firstly been a transformative driver in changing the way that sea turtle is used as a resource – from consumptive to a non-consumptive one, as presented in **Figure 2**.

It is important to recall that long before TTE was established, sea turtle eggs were traditionally a plentiful subsistence only common resource among coastal villagers in Terengganu. With the advent of modern transportation networks after the Second World War, which improved access to the market that were further away from nesting grounds, sea turtle eggs later became a traded commodity, i.e., for income resource as the egg surplus

could now be sold for cash. Henceforth, collecting and selling turtle eggs became a new livelihood option for the coastal folks. However, the popularity of this free-for-all trade resulted in disputes that were grave enough to compel a legal intervention from the state, i.e., the TTE establishment in 1951. As a result, sea turtle eggs that were previously a common resource became an "exclusive resource" – one that could only be accessed via commercial tender or special rights. But from the conservation perspective, this had little impact. As explained in the earlier subsection, the initial TTE was a trade-oriented legislation. Thus, all eggs collected then still went to the market, only through fewer hands as people had to earn the state's consent to do so either commercially via concession or politically via special rights. In response, within a decade of TTE, conservationists began setting up hatcheries in Rantau Abang – a sea turtle conservation hotspot due to the massive landing of leatherback turtles – with the support of the DoF that was then officially not part of the legal framework. To get the eggs for incubation, the egg buyback scheme was launched. The same approach was used when hatcheries were set up in late 1970s in Perhentian Island and Setiu, targeted to green and olive ridley turtles, respectively. Indeed, through the egg buyback program, the transformation of this natural consumptive capital into one that is non-consumptive slowly began by encouraging license holders to voluntarily support conservation by selling their collected eggs to hatcheries instead of the market.

The amendments in 1987 and 1989 enabled the setting up of turtle sanctuaries and put an end to the leatherback turtle egg trade, respectively, – further transforming the nature of this resource – at least for one species – from an exclusive resource to one that was no longer accessible. More importantly, it resulted in the appointment of the DoF as the Authorized Officer for the implementation of the conservation elements of the TTE. The DoF has since become the key agent in sea turtle management in Terengganu. The agency also contributed to raising public awareness on the importance of marine conservation in the early 1990s, in support of the newly established marine park islands in the state that was administered by their Federal counterpart. With increasing awareness raised on sea turtle conservation through the establishment of the Turtle Information Center in Rantau Abang, as well as the increasing popularity of Terengganu's marine protected areas where turtles are given Federal protection, sea turtle eggs became increasingly more popular as a natural heritage to conserve, instead of a natural resource to consume. Equipped with the necessary mandate in hand, DoF introduced more measures to protect important nesting beaches in the late 1990s, through the administrative conversion of more important rookeries into turtle reserve beaches – all selected among the beaches with the highest nesting density and particularly those with operating hatcheries. These reserve beaches are delisted from the tender list, i.e., no longer available for legal tender and became fully managed by the DoF where locally hired rangers (usually selected among egg collectors) are responsible for managing the sites. Administratively created, reserve beaches, however, are not turtle sanctuaries. Therefore, the authority of the DoF rangers is limited to accessing the nested eggs, similar to the license holders' at licensed beaches.

This paper has also found that although not amended since the past three decades, DoF has used TTE's provisions to administratively take the management of this natural resource to go beyond a coexistence between concession and conservation, to currently exist in practice in the form of a pro-conservation legal framework supported by the concession system. Here, "pro-conservation" refers to a system that is "more supportive toward" instead of "completely in agreement with" conservation motives. This move was achieved by the state DoF via the provision in Section 9 that made the sale of all collected eggs from licensed beaches to DoF for incubation compulsory in the terms and conditions of the tender since 2003. Through this addition, the state further prevented the sea turtle eggs from being traded, hence making a mandatory non-consumptive transformation of the concession system. This means that license holders are in effect working not for the market but for conservation. But such an ambitious move would have a huge financial impact on the state because it would mean repurchasing eggs from 36 nesting beaches. Instead, the agency had strategically delisted the most productive licensed beaches in the state, such as Chagar Hutang in 2005. By 2007, the TSAC has agreed to designate a new status to these most productive beaches as "turtle sanctuary" under the National Land Code (NLC), but not TTE. Meanwhile those that had low nesting record or moderate nesting but difficult to patrol were kept as licensed beaches, i.e., part of the tender list. In other words, DoF's mode of selection between licensed, reserve, or sanctuary beaches appears weighted on the cost implication from the egg buyback scheme: sanctuaries are created only at sites that would incur a significant cost to purchase all eggs due to high-density nesting but are practically easier to patrol, while low-density nesting beaches that are more cost effective to be leased were listed for the tender. Other beaches among licensed beaches that fall in between the two categories (moderate nesting and easy to patrol) were listed as reserve beaches instead. Through this decisive move, the state has done well beyond the suggested conservation target of protecting at least 70% of the nests (Chan, 2004), as shown in **Figure 3**.

Transformative Adaptations by Sea Turtle Trade-Dependent Livelihood Makers

It is important to note that in theory, these transformations through mandatory sale of eggs from licensed beaches as well as converting the most productive among them into NLC sanctuaries or reserves had little, if at all, livelihood implications on those involved in the trade. For egg collectors, the mandatory egg sale had no repercussion on them unless they were also license holders. This is because egg collectors who work as "Ranger Pemajak," i.e., for license holders (livelihood-dependent category) to supplement their low income will still get paid for their effort regardless of the status of the beach. In the case where a beach is converted into a reserve, they will most likely be recruited as rangers by DoF, which is also applicable to license holders who collect eggs on their own. The profiling of these various groups of concession participants that were interviewed as presented in **Table 2** provides good insights on this matter. Among the latter, a fraction of them are those who despite their financial hardship secure the tender to maintain access to their family's

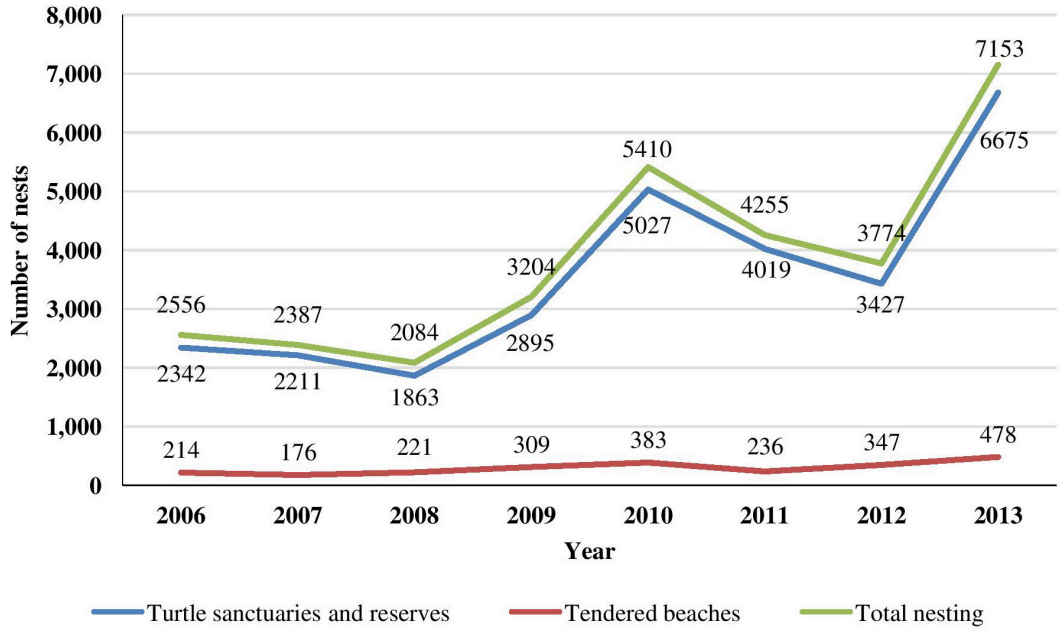


FIGURE 3 | Green turtle nesting record based on beach classification in Terengganu from 2006 until 2013.

"traditional nesting beaches" where sea turtle eggs have been collected for livelihood for generations, hence categorized as livelihood-dependent also. But generally for license holders, there is of course the potential loss of income when fewer beaches became available for tender. On this note, based on the relatively stable nesting number recorded at licensed beaches in stark contrast to the fluctuating trends recorded at turtle reserve and sanctuaries (see **Figure 3**), it could be suggested that license holders who are legally compelled to report the landing data on their licensed beach may underreport in an attempt to mask its real productivity, to prevent delisting of their licensed beach in the future, as opined three decades ago by Siow and Moll (1982). Losing a beach tender does appear to be a greater cause for concern than the mandatory sale that cuts them off the market trade, which surprisingly did not matter much to most of them as long as they get paid, even for a lower rate than the market. They – termed financially motivated concessionaires – explained that there are benefits in selling to hatcheries that outweigh selling to the market, a point to which we return in *Egg Buyback Scheme: A Protection Tool With Unfulfilled Potential.* The fact that license holders have been encouraged to voluntarily sell their eggs to hatcheries since the 1960s may also have helped to prepare this group for the pro-conservation move when it took effect. Furthermore, most of them hardly depend on the concession for their livelihood, which is already secured via other activities. Peculiarly, however, there are also non-financially motivated concessionaires, who may be the least supportive of the pro-conservation legal framework, due to their consumptive agenda for sea turtle eggs, a point that is also discussed further in *Egg Buyback Scheme: A Protection Tool With Unfulfilled Potential.* As for the egg sellers in the local markets, the opportunistic, non-exclusive nature of their trade makes them the least dependent on this resource, which they willingly sell when offered to them by those who have the supply, further claiming that they wouldn't if it was prohibited. For the same reason, the delisting of more beaches for protection has little impact on them. It is also found that the move toward non-consumptive use of sea turtles was gaining ground especially with the growth of the tourism industry in Terengganu where sea turtles are an icon. The data collected on the type of livelihood activities among the local communities before and after the conversion of main nesting beaches in the marine park islands of Redang and Perhentian into turtle sanctuaries under the NLC demonstrate this well. Prior to the establishment of these islands as a marine park in 1993, 86% of the interviewed households depended on fishing activities and egg collection for their livelihoods, whereas the remaining were involved in non-natural-resource-based livelihoods. Now, 44% of these households are either running a business, self-employed, or being hired in the tourism industry. Meanwhile, 14% work in conservation as marine park officers or DoF contract rangers, 35% are in non-natural-resource-based work such as government servants, and only 7% remain in the fisheries industry.

TTE's Gaps Between Paper and Practice

Earlier results demonstrated that TTE has provisions that enabled the practical transition toward non-consumptive utilization of sea turtle eggs by those involved in the system in Terengganu, through a legal framework that has evolved from a purely trade-oriented system to one that increasingly grew to support conservation, into becoming a pro-conservation concession system. Over time, the sea turtle eggs that were in the beginning of TTE a purely traded commodity became partially conserved via legislation through the creation of turtle sanctuaries as well as the trade ban of leatherback turtle eggs. Furthermore, it made

TABLE 2 | Categories of concession participants based on their profile and livelihood dependence.

Category	Profile	Livelihood dependence (on Concession)
"Ranger pemajak" (Employed egg collectors), *n* = 3	No stable income (coastal fishers) who work as egg collectors at night to supplement household income. Some would bid for the tender if they had the financial means while others would prefer working as DoF Ranger for fixed, stable income.	Yes
Traditional concessionaires, *n* = 3	No stable income (coastal fishers, one retired) but try hard to raise fund using saving, loan, or financial aid to secure tender to keep family's traditional livelihood practice of collecting eggs.	Yes
Business-minded concessionaires, *n* = 6	Have other stable source(s) of income but seek to increase wealth though turtle egg sale. Employ Rangers to patrol and collect eggs at their licensed beaches.	No
Pleasure and leisure motivated concessionaires, *n* = 4	Have other stable source(s) of income but seek exclusive access to nesting beach and eggs for (own/family/friends/colleagues) entertainment and consumption. Employ Rangers (usually family members) to patrol and collect eggs at their licensed beaches or do it themselves.	No
Market traders, *n* = 9	Opportunistic sellers (turtle eggs are one of many other products sold), only when offered by suppliers (concessionaires, random local people from Redang).	No

room for increasingly more conservation measures such as the egg buyback scheme and the status change of nesting beaches to be implemented. The former also involved conservation NGO as egg protection programs had become increasingly more popular since the launch of the first hatchery just a decade after TTE's inception. Through various strategies, both resource managers and users that have either concession or conservation motives have learned to coexist by responding accordingly to these gradual transformations to achieve their respective goals. But our study has also found that not all that was passed by the lawmakers three decades ago were put into practice. In this subsection, we present the important gaps that were found, particularly in relation to the governing institutions, as well as the unfulfilled potential of powerful protective provisions – all linked to the stakeholders' various capital profiles that influence their decision-making.

"Who's Who" in Terengganu Sea Turtle Management Scene: The Multitasker, the Helpful Allies, and the Absentees

Although the law has, on paper, provisions that could resolve most of the major enforcement issues highlighted by scientists and conservationists, our findings suggest that not all provisions were effectively put into practice. Indeed, DoF currently seems to have the most responsibility among all the organizations involved in the management of sea turtles in Terengganu (**Figure 4**). The amendments in TTE in 1987 were found to have structured the operationalization of its legal framework under three distinct functions, i.e., conservation, concession, and enforcement. However, DoF has two roles to play, as the Deputy Licensing Officer under Conservation and as an Authorized Officer under Enforcement. Hence, they manage turtle hatcheries and nesting beaches, and collect data and information related to sea turtles in Terengganu, as well as serve as the secretariat of the TSAC that advises the Terengganu State Executive Council on all matters related to turtles. With the pro-conservation concession scheme, there is at least no need for the agency to also take care of all the nesting beaches listed for tender. They only need to buyback collected eggs from these sites, where the responsibility of important tasks such as recording nesting data and patrolling the beaches against poachers is taken up by the license holders.

As for the 18 declared turtle sanctuaries and reserves, DoF currently fully manages 14 beaches and relies on assistance from their allies for the other four. These allies, such as the Sea Turtle Research Unit of Universiti Malaysia Terengganu (SEATRU-UMT) and World Wildlife Fund-Malaysia (WWF-Malaysia) that have been helping DoF's conservation work since the 1990s, are not part of the official TTE conservation structure. They are, however, part of the TSAC and have been DoF's most reliable partners, i.e., social capitals in the SLA terminology, that began raising funds to purchase sea turtle eggs at important nesting areas as well as managing hatcheries (see **Figure 4** in red). Only when Redang Island and Ma'Daerah beaches were gazetted as turtle sanctuaries under NLC in 2005 were they no longer needed to purchase the eggs to support the hatchery-related conservation work. Before that, SEATRU-UMT had been running the highly productive Chagar Hutang Beach since the 1990s and met their financial needs to purchase the abundant sea turtle eggs and manage their geographically remote project site at Redang Island through novel and innovative fund-raising measures such as "adopt-a-nest" and paid-volunteer programs (Chan, 2013). By the time the beach was converted into an NLC turtle sanctuary, SEATRU-UMT had raised funds to purchase more than 300,000 sea turtle eggs for incubation (Chan, 2004). WWF-Malaysia meanwhile has successfully gained financial support from private corporations as well as through public donations to realize their conservation efforts in the southern and northern nesting sites of the state. Other organizations such as the Lang Tengah Turtle Watch (LTTW) at Lang Tengah Island and Bubbles Dive Resort at Tanjung Tukah Beach, Perhentian Island have also joined in the effort. The former has boldly taken the challenge in competing with commercial license holders to acquire nesting beach tenders, becoming the first "for conservation" license holder since 2014. Meanwhile, Bubbles Dive Resort paid a donation to DoF to acquire a special license to patrol a turtle sanctuary at Perhentian Island.

The above-mentioned conservation allies have become an integral part of the co-management of sea turtles in Terengganu. However, the enforcement for these sites is beyond their jurisdiction as they are not officially authorized enforcers of TTE, except when they also hold a license like LTTW. Therefore, although the earlier discussed findings suggested that there is sufficient law and structure to ensure the effectiveness of TTE

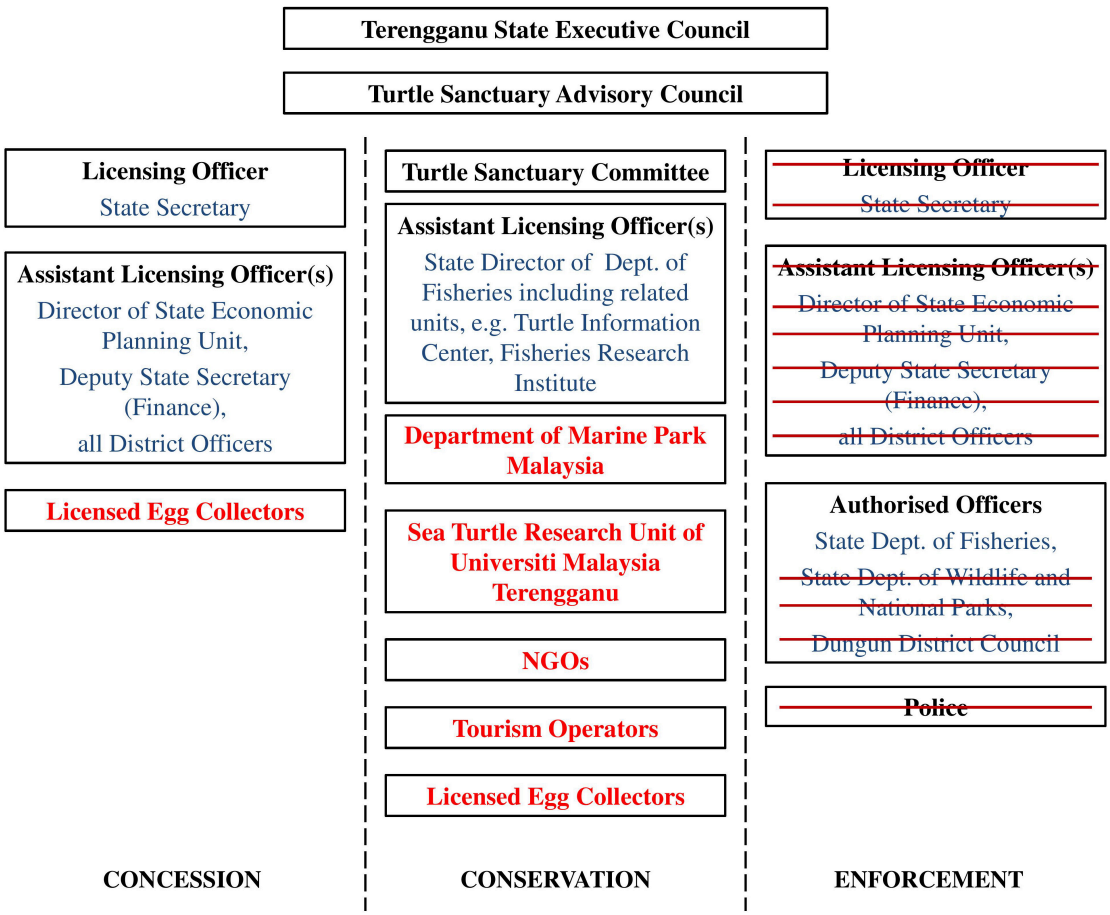


FIGURE 4 | Institutional framework for sea turtle management in Terengganu based on what is practiced in reality, showing the institutions and their appointees (in blue) according to their roles in sea turtle egg concession, conservation, and enforcement. Those crossed in red are structures or agencies that are in reality found absent or not carrying out their roles. Institutions (in red), not provided in the legal institutional, but are added for their role in sea turtle conservation.

where listed beaches are concerned, which would cover all important nesting areas in the state, enforcing the law is indeed a daunting task due to manpower shortage as per the testimony of interviewed DoF officers. However, this human capital deficit would not be as critical if all agencies given the enforcement authority played their roles, and not just the DoF. Institutional absenteeism is instead found: despite extensive provisions given to the Police to carry out enforcement work related to the enactment, there is no documentation of any enforcement activity carried out by them. Similarly, there is no record that the Licensing Officer, i.e., the Terengganu State Secretary, has directly played its enforcement role despite having the legal jurisdiction to do so. The Licensing Officer appears to continue playing their traditional role of managing the concession only.

Egg Buyback Scheme: A Protection Tool With Unfulfilled Potential

With the establishment of a pro-conservation legal framework via the mandatory egg buyback scheme, the egg concession has become a mechanism for conservation, i.e., a pro-conservation concession system. However, our review on its implementation suggests that its full potential was never fulfilled as it appears that DoF did not buy all the eggs from the licensed beaches. For example, in 2007, 422,000 eggs were estimated to have been traded (TRAFFIC Southeast Asia, 2009), which is almost double the total number of recorded nestings in Terengganu for that year (see **Figure 3**). Various challenges impede DoF's ability to fulfill its buying obligation, particularly the agency's lack of financial capital to buyback all the eggs. But it does not appear to be an insurmountable challenge as DoF could rely on the help of local NGOs. In 2006, WWF-Malaysia had proactively started to purchase turtle eggs from license holders to increase DoF's incubation rate. In Geliga, 1,300 nests were incubated in 2012 as compared to 300–400 nests prior to 2009 (WWF, 2014). Their participation in the scheme ensured that fewer number of eggs from licensed beaches would end up in the market. When fully implemented, the egg buyback scheme can lead to increased volume of incubated eggs in hatcheries as demonstrated by the data collected from WWF-Malaysia's report in 2014 to the TSAC in **Figure 3**. However, WWF-Malaysia stopped buying eggs since 2014, potentially resupplying the eggs to the local markets as the license holders who had financial interest in the

scheme resorted to cover their investment by selling all their eggs commercially instead. The withdrawal of WWF-Malaysia from the scheme in 2014 was a blow to this pro-conservation concession system that no longer could buffer for DoF's human and financial capital shortage.

Besides lack of funds, there are also other challenges in fully implementing the egg buyback system. The license holders had made complaints regarding the DoF bureaucracy in getting their payment after sending their eggs to the hatcheries. However, they admitted at the same time that DoF officers are sympathetic to their administrative woes and always came to their aid in expediting the payment. There were also concerns regarding the extra care that needs to be given to eggs that are transferred to the hatchery for incubation to ensure the survival of the embryo unlike eggs sent to the market that would be consumed anyway. However, under the WWF buyback program, there was an additional financial incentive given to license holder for each successfully hatched egg that compensated for the technical hassle – easily avoided should *in situ* incubation be practiced. Even so, not all license holders sell their eggs to DoF because the concession also attracts those who seek to personally have exclusive access to egg collection for their own consumption and/or to be given away as rare gifts. This applies to elite license holders including businessmen, government servants, and politicians whose livelihood is not at all dependent on the egg collection, hence irrelevant to the objective of the concession anyway.

Finally, this study found that the local demand for sea turtle eggs remains strong. Interviews with local market traders who all sell them not as a main product but one of many also suggested that the demand remains healthy, even necessitating the smuggling of eggs from Sabah and beyond when there is limited local supply. Apparently, buying turtle eggs at the local markets has also become a tourism activity in Terengganu. According to a state DoF officer, only 10% of eggs sold in Terengganu are sourced from licensed beaches while the other 90% are smuggled in from neighboring countries (Sim, 2015). But such demand for eggs need not deter the implementation of the scheme. This is because for these "gourmet" consumers, the eggs are a delicacy and not staple food. Hence, they would accept its market rarity should there be much less supply. As the mandatory sale to DoF is limited to the licensed beaches only, the supply of eggs to meet local demand can be sourced among eggs that could legally be collected from a number of non-listed beaches in the state where turtle nesting does occur, albeit rarely. All the challenges and opportunities mentioned above are presented in **Table 3**.

TABLE 3 | Challenges and opportunities in the implementation of the mandatory egg buyback scheme.

Category	Challenges	Opportunities
Financial	DoF has insufficient fund to purchase all collected eggs from licensed beaches or match market price.	License holders prefer to sell to DoF at even lower price because DoF in theory would buy all, while traders only buy the quantity they need. Fundraising for egg buyback is a publicly well-supported activity that can be supported by NGOs.
Administrative	Claiming payment can be a hassle for license holders who do not get paid immediately while the market pays cash on delivery.	DoF is perceived to be quite understanding of their license holders' constraints in meeting documentation needs.
Technical	Collected eggs must be handled carefully during transfer to hatchery (when not for *in situ* incubation) to ensure incubation success, which is not applicable for market-destined eggs.	Financial incentive is given to license holders for each hatchling recorded, in addition to the payment for the eggs. Only applies to *ex situ* incubation.
Personal agenda	Some license holders seek to secure the eggs for own consumption or as rare gifts to be given away (i.e., social capital), so egg buyback will never be an option.	The concession is meant to support turtle egg-dependent local livelihood. This minority group should be excluded from the concession due to their livelihood security.
Market demand	Turtle eggs remain a sought after delicacy by locals and are now becoming a new tourism attraction, bought by curious tourists from other states.	A full implementation of egg buyback will not affect non-listed beaches where less frequent nesting occurs. Eggs collected there could be traded, although less abundantly. But as a delicacy, its market rarity will be tolerated.

Almost but Not Quite TTE Protection for Nesting Sites

As discussed above, TTE is indeed not a full-proof legislation for sea turtle conservation, not because it allows for concession to exist but because the provision that protects sea turtle eggs from being traded under Section 7 is applicable to listed beaches only. Sea turtle egg incubation statistics (see **Figure 3**) do suggest that DoF has always maintained the strategy of ensuring at least 80% of deposited eggs are incubated annually as its main conservation approach – which is well above the 70% mark suggested by scientists (see Limpus, 1993; Chan, 2004). It has successfully done so by protecting all the high-density nesting beaches under various protection schemes (see **Table 4**), while the less productive ones are either listed for commercial tender or unlisted. However, the more recent sanctuaries were not established under TTE but under NLC. Some of the former are located in the state's marine protected areas, which used to be under the "one island, one tender" approach prior to 2006. Under this scheme, all beaches were listed under a single tender for each island. In 2006, only a few nesting beaches that fulfilled specific criterion – remote, short bays with the most productive nesting record such as Tiga Ruang Beach, Pinang Seribu Beach, Tanjung Tukah Beach, and Tanjung Guntung Beach in Perhentian Island, as well as Chagar Hutang Beach, Che Keling Beach, Bujang Beach, Mak Kepit Beach, and Mak Simpan Beach in Redang Island – were selected as turtle sanctuaries under NLC – a hybrid sea turtle habitat protection scheme. The other beaches, however, became unlisted – not in the tender list nor given any conservation status.

This move was perhaps to make way for the NLC beaches eventual gazettement as sanctuaries under the TTE – the most

TABLE 4 | List of nesting beaches classified as TTE/NLC turtle sanctuaries or turtle reserves in Terengganu.

District	TTE turtle sanctuary	NLC turtle sanctuary	Turtle reserve
Dungun	Rantau Abang Beach	–	Rantau Abang 1 Beach Rantau Abang 2 Beach
Kemaman	–	Ma'Daerah Beach	Geliga Beach Rhu Kudung Beach Chagar Hutan Beach
Setiu	–	–	Telaga Papan 1 Beach Telaga Papan 2 Beach Kuala Bharu Selatan Beach
Besut	–	Perhentian Islands: Pinang Seribu Beach; Tiga Ruang Beach; Tanjung Tukah Beach; Tiga Ruang 2 Beach; Tanjung Guntung Beach	–
Kuala Terengganu	–	Redang Island: Chagar Hutang Beach; Mak Kepit Beach, Bujang Beach, Mak Simpan Beach; Che Keling Beach	

comprehensive protection status that puts everything from land development to turtle nesting matters under the same jurisdiction. Under TTE, the turtle sanctuary beach land is protected exclusively for turtle nesting, where no egg collection is allowed and the area is out of bounds for the public during certain hours to ensure optimal condition for nesting turtles. However, this has yet to happen after more than a decade and Rantau Abang remains the only TTE turtle sanctuary since 1989. As for these other "hybrid" sanctuaries, although gazetted under NLC as turtle sanctuaries, they are only protected from development by the District Office and do not necessarily have access to the protective provisions under TTE unless officially declared as listed for conservation by its authority. While their land matters legally fall directly under the District Officer, matters related to turtle nesting are assumed by various organizations, i.e., SEATRU-UMT in Chagar Hutang, WWF-Malaysia in Ma'Daerah, and DoF in the rest. Meanwhile, turtle reserves, the third type of listed beaches, are managed by DoF, whose presence on site provides a protection as sound as TTE sanctuaries, although they technically are no different than licensed beaches, except that DoF is the operator instead of a license holder. Fortunately, the ambiguous status of NLC turtle sanctuaries and DoF-operated turtle reserves never seemed to be questioned, enabling total conservation. But, eggs from the rest of the beaches in these marine park islands – now all unlisted – could end up being traded in the markets. Furthermore, the local sea turtle egg traders openly list the exotic (MPA) origin of their Redang sourced goods – much to the consternation of conservationist.

Other Issue: Terengganu as a Turtle Egg Trading Hub

Besides the presence of "Redang" eggs in the local markets, there are also those that presumably hail from other states, particularly Sabah where a total trade ban has been declared. This is because TTE does not prevent eggs sourced from other states and countries from being traded in Terengganu. This strikes another blow to Terengganu's conservation image, denounced as a hub for illegally sourced eggs from another state and neighboring countries (The Star, 2006, 2010, 2015; Sim, 2015). The local traders claim to be not inclined to market them with a "Terengganu" label to avoid ruining one's reputation by selling these externally sourced eggs as local eggs due to their reportedly inferior quality compared to the locally sourced ones, which local consumers apparently can easily detect and prefer to avoid. These eggs are usually bought by less knowledgeable, curious tourists from other states in Malaysia or foreigners instead, some of whom purposely seek the eggs at the local market.

DISCUSSION

By examining the conditions and consequences of the Terengganu sea turtle legal framework, the paper revealed the interdependencies between the access component (what policy is put in place) with those of stakeholders' strategies (what they do in response to the policy) using their available assets (the various resources that they have), which resulted in the current conservation and livelihood outcomes. The findings made by reading the enactment carefully do imply that in theory, there are provisions to ensure that conservation thrives in the state despite its coexistence with the sea turtle egg concession. This is because what has been legally put in place in this pro-conservation legal framework is in fact a concession system where even the commercial egg collectors now work for conservation. This scheme is unlike the withdrawal right given to the local communities in Ostional, Costa Rica to collect the eggs as prescribed by the law during the mass synchronized sea turtle nesting called "arribadas" for trade or own consumption (Campbell, 2007), which is more comparable to the case studied in Venezuela by Barrios-Garrido et al. (2017). Instead, it is quite a unique model because in Terengganu's case, the Licensing Officers used their management rights to reroute the eggs away from the market to the hatcheries. Some, however, would argue that the "commodification" of the eggs, i.e., payment to the license holders for the eggs, makes it unworthy for conservation because it is "philosophically suspect" (Ferraro, 2007: 32). It conflicted WWF-Malaysia's policy (Zolkepli, 2012), which, although acknowledged the Malaysian egg buyback program as a success, alleged that it "still puts a commercial value on turtle eggs instead of a conservation value" (WWF, 2014:11). Often, moral, instead of biological, reasoning does lead sea turtle experts and conservationists to denounce any legal exception to a trade ban even when it improves support for local conservation (Campbell, 2007). The repeated calls for TTE to include a total ban on all sea turtle egg trade as the recommended solution for weaknesses found in the management of the species (Bernama, 2006;

TRAFFIC Southeast Asia, 2009; WWF-Malaysia, 2012, 2018; Zolkepli, 2012) are no different, recommending the moratorium to change local consumptive behavior. Their concerns are indeed valid as Terengganu – synonymous with sea turtles since the heydays of leatherback turtle landings in Rantau Abang – must learn the important lessons from the trade ban on leatherback turtle eggs in 1989 that had, however, come too late, and failed to avoid the extinction of the species (Liew, 2011). For this reason, sea turtle conservation is a matter of concern in Terengganu as well as in the Southeast Asian region. Hence, the labeling of this scheme as pro-conservation may indeed face fierce criticism as TTE does not currently cover all beaches in the state, only listed ones where nesting density is substantial. Therefore, there is still open access to local sea turtle eggs found and collected at random beaches that may still potentially end up in the market, traded among extra-Terengganu eggs as explained earlier. But there is perhaps wisdom in this approach of compromise. After surveying the various traditional consumptive uses of sea turtles in Southwestern Africa, Fretey et al. (2007) cautioned that conservationist must respect local traditional use – including consumptive in nature – if they wish to save sea turtles because in such localities, legal prohibition imposed in the name of species conservation alone rarely works. It is a reality corroborated by IOSEA's 2014 report on the unlawful use of turtle parts in 75% of its member countries. Therefore, scientists and conservationists, often the privileged rather than marginalized group in policy-making (Guha, 2003), must reflect on the political correctness of their motivations in overriding local mores in the name of sea turtles. Demands are now being made (Rudrud, 2010) or defended (Barrios-Garrido et al., 2017) by indigenous peoples on their customary rights to consume sea turtles, and in places where human consumption has long been part of traditional but effective local ecosystem management strategies, turning a deaf ear is nothing more than an emotional, evasive stand. But Campbell (2007) has shown that heeding local claims to consumptive use is perhaps counterintuitive among "for turtle" folks. Her work in Costa Rica has shown how some turtle conservationists have resorted to using more serious regional or global endangered status to defend or push local conservation agenda, although the local population of the species is actually not in grave danger. This is why this paper maintains that TTE in its current form is unique because it has shown an ingenuity of seeking balance between the often conflicting priorities of local people and conservationists. It exemplifies a management strategy that is scientifically guided without losing its intuitions for local traditions and practices. Originally established in 1951 exclusively for trade regulation, it has gradually become a policy that is in fact biased toward protection, i.e., pro-conservation. Over time, TTE has what it takes to make it possible to achieve the desired outcomes of sustaining the Terengganu sea turtle populations and turtle-egg-dependent local livelihoods should all the legal provisions be optimally put to use in practice.

The results do show that this unique pro-conservation legal system has been the driver for a progressive shift from consumptive, i.e., for concession, to non-consumptive, i.e., for conservation, utilization of sea turtles as a resource through various strategies that the multiple actors in the policy realm of sea turtle management have put into practice toward gaining the best outcome they could expect with the resources that they have at their disposal. But as highlighted in the section *TTE's Gaps Between Paper and Practice*, there are gaps in its implementation. Nonetheless, the current management strategies carried out are a result of maximizing strengths in certain capitals for compensating limitations in others in order to achieve a balanced outcome between supporting the livelihood of those dependent on the trade and the sustainability of sea turtle population. The DoF has therefore enhanced the use of the egg buyback scheme as an effective management tool by embedding the potential cost of egg buyback in their selection criteria for licensed beaches. For this reason, the paper differs in opinion with views that the egg buyback scheme should not be encouraged (Kent, 2006; The Star, 2011). It argues that the egg buyback scheme is a perfect companion to the conservation approach currently taken in Terengganu because it effectively helps DoF to cope with the human capital constraints that it faces through the optimal use of the same capital that the license holders readily have access to: egg collectors who are able bodied and possess valuable local ecological knowledge on sea turtle nesting behavior. The former, due to the long tradition and practice of egg harvesting in the state, are highly knowledgeable and skilled in finding sea turtle nests and are in fact the real experts in the field (TRAFFIC Southeast Asia, 2009). They, however, may not all want to be employed as rangers due to the seasonal nature of the occupation and low wage, which is just one of the many livelihood strategies that the coastal communities employ to achieve their desired livelihood outcomes. Therefore, ideally, the concession system provides a way to protect more nesting beaches with a limited budget, e.g., reducing the cost by buying eggs instead of paying for seasonal rangers in low-density nesting sites. Indeed, in many places where sea turtle conservation has livelihood implication, using such economic incentives, although philosophically troubling to conservationists, is the most effective way to ensure that conservation goals are met (Ferraro and Gjertsen, 2009). Moreover, in a true display of livelihood resilience, the local communities at the marine park islands that formerly used to co-own the tender through cooperatives have now ventured into tourism operations, where sea turtle-related tourism (i.e., turtle watching in the water) is now the most popular attraction, similar to the leatherbacks in the past. The spill-over effect from sea turtle tourism is enjoyed by local traders whose various turtle inspired merchandises are coveted by tourists and locals alike. Hence, sea turtles are reclaimed as a natural capital by the Terengganu people, in this context more valuable alive than consumed, which is similar to Tortuguero where turtle tourism has become a major source of revenue to the local communities (Place, 1991). This option, recommended by conservationists (Sardeshpande and MacMillan, 2019), must, however, be thoroughly evaluated and strategically incorporated into the legal framework to avoid abuses. But in sum, we opine that the current management of sea turtles in Terengganu is a resilient "making do" model of compromise that would still result in (i) the protection of the most important nesting sites to ensure the incubation of at least 70% of the nests in sustaining the local sea turtle

population while (ii) sustaining egg-collection-dependent local community livelihood via employment at reserve beaches and sanctuaries as conservation rangers or at licensed beaches as commercial egg collectors.

But is this outcome – one that results from maintaining the *status quo*, i.e., a quasi-practiced pro conservation legal framework – enough? To this, our short answer is yes, if the state of Terengganu is set to only maintain the population via incubation of at least 70% of the nests as recommended by scientists. However, it must be willing to face continued criticism for not committing to a legal condition that it has introduced, i.e., the egg buyback scheme, as well as for not doing enough enforcement, thus allowing Terengganu to become a sea turtle egg trading hub in the region (The Star, 2015; WWF-Malaysia, 2018). Our findings on the existing discrepancy between paper and practice have shown that more could already be achieved by doing a little more, i.e., implementing the pro-conservation concession system in full without even making any change to the law. The non-optimization of the law must be understood as the root cause for the setbacks against achieving the current management goal of the sea turtle egg trade, and not the lack of law. We do therefore opine that it is crucial to immediately address the challenging issues that hamper the fulfilment of TTE's full potential. After all, legislation is naught without effective implementation (Koch et al., 2006; Mancini et al., 2011). Hence, there would be no point to consider more law to protect sea turtles in the state if maximum effort was never given to fully put the existing one to optimal use. To begin with, the "institutional absenteeism," which we highlighted in the section *"Who's Who" in Terengganu Sea Turtle Management Scene: The Multitasker, the Helpful Allies, and the Absentees*, could be immediately remedied by increasing the presence of enforcement through the active participation of the police force. Their absenteeism compromises the optimal operationalization of this important function because poaching becomes a serious problem when enforcement is perceived to be slack (Keane et al., 2008). Simply due to their omnipresence and their status as a publicly recognized enforcement authority, members of the police corps would positively impact the lawfulness of those involved in not only the trade but all activities related to sea turtles including conservation and tourism in accordance to TTE legal provisions without increasing the number of DoF personnel. The Licensing Officer could also involve other well-meaning, local individuals or groups with proven good record such as MEKAR and PEWANIS – community-based conservation groups set up with the help of WWF Malaysia – by issuing special permits to help with enforcement at nesting beaches, from simple patrolling to even investigations at specific listed beaches. This would help increase the local communities' role in co-managing sea turtles, which Shyuji (2015) found was lacking.

Although the egg buyback scheme would have ensured the total incubation of all sea turtle eggs from licensed beaches, DoF, in our opinion, was perhaps never able to fully commit to it due to the colossal financial implication it would incur. Having largely achieved the "conservation quota" as discussed earlier in the section *TTE in Practice: Transforming Sea Turtles Eggs From a Consumptive to a Non-consumptive Resource* through the sanctuaries and reserves, the balance sold to the market is perhaps deemed insignificant. However, the mandatory contractual condition has been pragmatically maintained to this day, presumably as a useful backup strategy that can be used when nesting rates are low, requiring the acquisition of sea turtle eggs from the licensed beaches to ensure meeting the said target. But when not purchased for conservation, a gray market for the non-purchased eggs is created, where although illegal in theory, DoF would not be in the moral, if not legal, position to take action against their sale as the agency has not fulfilled its end of the contract. This is the reason why all the enabling factors for a fully implemented pro-conservation concession scheme, i.e., the opportunities presented in **Table 3** that could address the challenges in implementing the egg buyback scheme, must be put to good use. This means that the state must increase their support to enable DoF to buyback all the nests from license holders, perhaps resorting to crowd funding mechanisms with the help of NGOs. In doing so, the access to trade will also be limited to almost null because eggs from all licensed beaches – i.e., nesting beaches identified for tender – will now be incubated as those from turtle sanctuaries or reserves. Meanwhile, the conservation activities such as recording and reporting nesting data as well as preventing poaching through beach patrolling that would have to be carried out by DoF will be assured by the license holders. As argued by Ferraro and Gjertsen (2009) based on a global review of incentives for sea turtle conservation, the egg buyback is the most economically cost effective and possibly most locally favored solution for sea turtle conservation when local harvesting is legally regulated. If all eggs from licensed beaches are bought back for conservation, there will only be those from non-listed beaches, a potentially insignificant amount. Other issues related to the egg buyback scheme must also be taken care of: simplified claim procedure could be introduced to facilitate repayment to license holders while good incentives such as the hatchling bonus should be maintained; more *in situ* incubation to reduce transfer hassle and increase incubation success; the bidding open only to real "professionals," i.e. those with non-consumptive (for financial or conservation gain) motives only, while those with consumptive agenda can join members of the public in purchasing legal eggs – sourced from non-listed beaches in the state or beyond – at the local markets. With these optimization measures, the outcome could be (i) better enforcement capabilities by DoF due to the increased human capital that can now fully play their role with a free conscience, having now fulfilled their end of the tender contract; and (ii) total absence of eggs from listed beaches at local markets.

The fully implemented pro-conservation legal framework offers, in our opinion, a better management outcome than the former. But it still means that some eggs nested in Terengganu can end up in the market – not yet an acceptable outcome in the eyes of some who wish to see the end of the trade. Indeed, the state could upgrade the current pro-conservation legal framework to a full-conservation concession system, i.e., the TSAC can, using the administrative means provided in Section 9, include all beaches that are currently not listed as turtle sanctuaries or reserves in the state as licensed beaches from where all the eggs collected would be incubated as well due to the mandatory sale condition

of the tender contract. However, DoF must be provided with sufficient financial support to buy all the eggs collected by license holders. To the authors of the paper, this is the preferred option at the moment for Terengganu should the state seek to conserve all locally nested eggs without disrupting the balance between livelihood and conservation. It is reiterated that the egg buyback scheme does not encourage trade but serves only as a buffer for turtle egg-dependent livelihood, as well as creating a cost-effective alternative management solution for nesting beaches. Meanwhile any egg collected from the non-bidden locations will still be legally protected and could not be harvested without the concession. This facilitates enforcement work as action can be taken on anyone else found to have sea turtle eggs in their possession because this administrative measure ensures that no Terengganu sea turtle egg should end up in the market, and those that do can be confiscated. This immediately allows taking action on the local eggs such as those contentious "Redang" eggs.

However, under this full conservation legal framework, nothing could still be done to eggs proven to hail from other Malaysian states, creating a nagging concern of the state becoming a trade hub for "extra-Terengganu" turtle eggs. Therefore, even the full conservation type of concession in Terengganu does not prevent the state from becoming a trade hub as only Terengganu eggs are prohibited from the market, but not those sourced outside of Terengganu. Despite a total ban in Sabah, illegal trade does take place discreetly in local markets, and high state officials have been caught red-handed for sea turtle egg fine-dining (Today Online, 2015). A huge quantity of "Sabah" labeled eggs end up sold in Terengganu, at a lower price than the local sea turtle eggs, as they arrive less fresh and have to usually be pickled, having traveled across the South China Sea to get to Terengganu (WWF-Malaysia, 2018). This is an important lesson to note, because the trade shall persist for as long as there is a demand (TRAFFIC Southeast Asia, 2009), which our findings have shown to be strong among the local population in Terengganu. It is also not impossible that these "Sabah" eggs do hail from foreign origins – but are being imported via Sabah (WWF-Malaysia, 2018), and labeled as such due to CITES. Indeed, while importing sea turtle eggs from any foreign country is illegal, there is no prohibition against selling eggs from Sabah elsewhere in Malaysia where the trade is legal. Perhaps a local version of CITES may be considered as a possible solution to this problem. Indeed, if the "Sabah" sea turtle eggs could be confiscated, a huge amount would disappear from the local markets. The origin of local sea turtle eggs supply could be verified through investigation by the local enforcement team, which would become better capacitated to do so with the full participation of all TTE nominated enforcement agencies as discussed earlier. Of course, the other solution to stop extra-Terengganu eggs that could also be taken is for the state to move toward amending the law – the first in almost 30 years – to impose a sea turtle egg trade moratorium on all species. This would turn TTE into a total conservation legal framework.

Indeed, the current conservation efforts in Terengganu, which include protecting turtle sanctuaries at nesting beaches and offshore habitats as well as providing education to increase knowledge and public awareness, may not be sufficient in sustaining a viable turtle population, considering the increased threats to the survival of this migratory species at various life history stages (Klein et al., 2017). It therefore may seem that since the egg buyback scheme is not sustainable without funding, a ban on the commercial sale and trade of sea turtle eggs may discourage their trade and increase the number of eggs incubated in the hatchery. But the paper affirms that a moratorium of the trade now may not likely solve the problem, as proven by the transgressions that have been recorded in Sabah (Sario, 2016; Kasmir, 2017). DoF is surely aware of the implementation challenges that a trade ban will have without popular public support. Indeed, sea turtle conservationists in Terengganu must gain a better understanding on the consumptive value of sea turtle among the local community before pushing for increased protective measures such as a total trade ban. It is also worth noting that when harvesting has always been part of the nesting equation, its absence could also be detrimental to the local sea turtle nesting population. Ferraro and Gjertsen (2009) reported how a total ban in an important but isolated rookery in Kalimantan resulted in more nest loss due to predation by rats that had flourished since the egg collectors ceased to visit the beach. On the contrary, their presence could help conservation work in the long run. Campbell et al. (2007) also noted that since the legalization of harvesting during arribadas in Ostional (Costa Rica), illegal harvesting has become less accepted by the local community, and better infrastructure has been provided by the community themselves to help patrolling activities. Furthermore, total conservation may even lead to sea turtle overpopulation that has been found to be damaging to local marine ecosystems (Lal et al., 2010).

CONCLUSION

By adopting the SLA approach, this study has found that while the TTE may not impose a trade moratorium yet, it still has powerful elements that are necessary in making sea turtle conservation a success since its transformation from a concession-oriented legislation in the 1950s to one that is pro-conservation in practice in the recent years. This shift has indeed made sea turtle conservation the state's priority over trade and enabled the incubation of enough sea turtle eggs for ensuring the sustainability of the sea turtles in the state as well as protecting genuine turtle-dependent livelihood. However, should turtle egg trade moratorium be the desired outcome for Terengganu, then the total conservation legal framework via trade ban will need to be supported by interstate legal mechanism as well, to ensure that the eggs will not simply end up being exported to other Malaysian states such as Kelantan and Pahang where demand also exists. It is therefore important to emphasize that this paper is not against imposing a law to ban the trade of sea turtle eggs, which should indisputably be the ultimate outcome in sea turtle conservation in Terengganu. However, legislative reviews take time, much more than administrative amendments such as the one that concerns the listed beaches. Furthermore, as discussed above, the full potential of the existing law that was diligently put in place with scientific insights and local sensitivity by lawmakers almost three

decades ago has not yet even been reached. It may therefore be more productive and realistic to now concentrate on optimizing what are already in place instead of asking for even more law that may only create new enforcement challenges in addition to those already discussed above. Limited enforcement capabilities coupled with strong local demand for egg consumption due to low conservation awareness will backfire against such a ban. Hence, efforts must be made in solving the problems related to the access to sea turtle conservation as discussed above to ensure the sustainability of both sea turtles and local livelihoods in the state's current legislation system. By turning it first into a full conservation legal framework, the state will pave the way for a total conservation legal framework via a total trade ban – to be implemented when all is ready to make a moratorium fully work.

ETHICS STATEMENT

This study was carried out in accordance with the recommendations of UMT Field Research Guideline, UMT Research Management Centre with written informed consent from all subjects. All subjects gave written informed consent in accordance with the Declaration of Helsinki. The protocol was approved by the UMT Research Management Centre.

AUTHOR CONTRIBUTIONS

JM, MJ, and SL contributed in gathering and analyzing the data, interpreting the results, producing the figures and tables, and writing the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00762/full#supplementary-material

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Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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frontiers in Marine Science

BRIEF RESEARCH REPORT
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Coral Diversity at Losin Pinnacle, an Offshore Reef in the Gulf of Thailand: Toward a Future MPA

Mathinee Yucharoen[1,2], Anchalee Chankong[3], Niphon Phongsuwan[3], Santi Ninwat[3], Udomsak Darumas[4], Suthira Thongkao[5] and Supaporn Phasombun[4]*

[1] Marine and Coastal Resources Institute, Faculty of Environmental Management, Prince of Songkla University, Hat Yai, Thailand, [2] Coastal Oceanography and Climate Change Research Center, Prince of Songkla University, Hatyai, Thailand, [3] Department of Marine and Coastal Resources, Ministry of Natural Resources and Environment, Bangkok, Thailand, [4] School of Science, Walailak University, Nakhon Si Thammarat, Thailand, [5] School of Engineering and Resources, Walailak University, Nakhon Si Thammarat, Thailand

Severe coral bleaching events in the Gulf of Thailand and along the Andaman Sea coast of Thailand caused widespread coral mortality in 1998 and 2010. The consequent decrease in coral populations impacted the structure, health, and services of Thai coral reefs. However, most colonies in the offshore reef of Losin were still alive after the coral bleaching events. Therefore, this study was conducted by the Department of Marine and Coastal Resources in order to help to establish a proposal for making it a Marine Protected Area (MPA). Surveys on coral diversity were conducted to produce a checklist of reef-building corals. Seventy-six coral species were found, with the most dominant species being *Porites lutea* and *Acropora* communities, such as *A. intermedia*, *A. grandis*, *A. muricata*, *A. cytherea*, and *A. valenciennesi*. This area is expected to be designated as a restricted MPA area, under the "Act on the Promotion of Marine and Coastal Resources Management B.E. 2558 (2015)." The high diversity of hard corals discovered in this study assists in promoting an Announcement of the Losin Marine and Coastal Resources Protected Areas following Ministerial Regulation.

Keywords: *Acropora*, coral bleaching, protected area, management, Gulf of Thailand

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****Correspondence:***
Mathinee Yucharoen
mathinee_yucharoen@hotmail.com

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INTRODUCTION

Coral reefs are one of the most vulnerable marine ecosystems to elevated sea temperature, and this has resulted in global coral bleaching (Moss et al., 2010; Hoegh-Guldberg, 2011; Burke et al., 2012; Kennedy et al., 2013; Graham et al., 2015; Hughes et al., 2017; Gintert et al., 2018). Predictions of annual coral bleaching occurring in the next 30 years are alarming due to the likelihood of increased frequency and severity (Manzello, 2015; Hughes et al., 2017). Increased coral bleaching and thermal stress impact hard coral species differently, with some species suffering significantly more mortality than others (Guest et al., 2012; Wooldridge, 2014). Thermal stress has not only been a concern for susceptible corals but is also a threat to coral reef health in terms of coral diseases (Gintert et al., 2018). Temperature-stressed corals could be more susceptible to opportunistic pathogens, which may be associated with subsequent diseases (Precht et al., 2016; Raymundo et al., 2018).

Widespread coral mortality from mass coral bleaching events between 1998 and 2010 in Thai waters has been reported previously (Yeemin et al., 2009; Phongsuwan and Chansang, 2012;

Sutthacheep et al., 2013; Yucharoen et al., 2015). Coral reefs along the Gulf of Thailand experienced high mortality, and a subsequent loss of *Acropora* species was reported in the region (Yeemin et al., 2009; Hoeksema et al., 2013). High losses of *Acropora* corals from these reefs were alarming, as some *Acropora* species could face functional extinction in the eastern Gulf of Thailand (Yeemin et al., 2013b). Additionally, the rapid growth of tourism, meaning an increased number of people visiting the reefs, led to additional physical damage, resulting in the temporary closure of dive sites, especially those in Marine National Parks (Yeemin, 2012).

The offshore reef at Losin Pinnacle appeared to be only minorly impacted by the mass coral bleaching that occurred in 2010, with relatively high live coral cover after the bleaching event. After the third global bleaching event in 2016, Sutthacheep et al. (2019) noted a decrease in bleaching relative to past bleaching events, with only 5% of the corals bleached. This mild bleaching could be due to its local conditions: surrounded by open sea, with strong currents and internal waves, which could dampen the impacts of increased temperature (Williams et al., 2010; Schmidt et al., 2016). The fishing activity around Losin Pinnacle may be an additional source of damage, as ghost fishing nets and gear have recently been found covering parts of the reef (Marine and Coastal Resources Research and Development Institute, 2018). The goal of this study was to supply the Ministry of Natural Resources and Environment with adequate information on the coral diversity and abundance found at Losin Pinnacle to establish baseline values for the reef in support of a marine protected area (MPA) proposal.

MATERIALS AND METHODS

Losin Pinnacle (**Figure 1**) is used as a base for lighthouse operations approximately 72 km from shore and is also known as one of the most southern popular dive sites in the Gulf of Thailand (844 SCUBA dives were recorded in 2017). This area is dominated by annual southwest and northeast monsoons during May–October and November–February, respectively. The wind speed toward the southwest direction is mostly higher than in other directions (**Supplementary Table S1**). An islet area is slightly above sea level, and the sea had a very small tidal range of about 50 cm (**Supplementary Table S2**). Surveys were conducted between 2015 and 2017 at six locations surrounding the reef. Belt transects, 30 m long × 2 m wide, with three replicates, were set parallel to the shore of the pinnacle in an anticlockwise direction at ST1–ST6 (**Figure 1**). Sites ST1, ST3, and ST5 were fixed at 20 m in depth for deep transects, whereas ST2, ST4, and ST6 were fixed at 10 m in depth for shallow transects. Underwater photographs ($n = 60$) were taken along each transect for assessment of benthic percentage cover. All images were analyzed by Coral Point Count with Excel extensions, using CPCe software (Kohler and Gill, 2006), and the benthic percentage was calculated. Coral species were recorded using a combination of belt transects and roving driver survey outside the belt transects in order to better assess true coral species diversity. The roving searches were done in shallow and deep areas, near the belt transects, covering 10–15 m and 20–25 m in depth, respectively. Coral photographs were taken, and some coral samples were collected for microscopic investigation when the researchers could not identify them *in situ*. All species were identified by morphological characteristics according to http://www.coralsoftheworld.org and others (Veron and Pichon, 1976, 1980, 1982; Veron et al., 1977; Veron and Wallace, 1984; Hoeksema, 1989; Wallace, 1999; Veron, 2000; Wallace et al., 2012). Genetic identification was not used in this study due to cost limitations. In addition, environmental parameters were measured *in situ* at sites ST1, ST3, ST5, and ST7 during August 28–30, 2017. The seawater temperature, salinity, pH, and dissolved oxygen data were recorded at 5 m depth from the sea surface by YSI Pro Plus Multi-Parameter Water Quality Meter (YSI Incorporated, Xylem Inc.), and the visibility distance was measured by Secchi disk (**Supplementary Table S3**).

RESULTS

Losin Pinnacle consists of a reef area encompassing about 65,000 m^2 and is associated with good water quality. The reef compositions on the south and west sides were different from on the east due to different levels of exposure to waves. As shown in **Figure 2**, it is clear that sites ST1–ST4 had high live coral coverage; 97%, 85%, 77%, and 95%, respectively. Dense *Acropora* communities at those sites included *A. intermedia*, *A. grandis*, *A. hyacinthus*, and several corymbose species. Some *Montipora*, *Porites*, and *Pocillopora* were found interspersed. Live corals at sites ST5 and ST6 showed lower percentage coverage, at 45% and 52% live hard coral cover, respectively. This windward side was dominated by large colonies of *Porites*, *Pocillopora*, and some branching *Acropora*.

A total of 76 hard coral species (**Supplementary Data Sheet 1**) were found, with *Acropora* species representing the largest number of species. The most dominant *Acropora* species were the arborescent, i.e., *A. grandis*, *A. intermedia*, and *A. muricata*, and the tabulate forms, i.e., *A. cytherea*, and *A. solitaryensis*. The other common *Acropora* species were of various forms; tabulate (*A. hyacinthus*, *A. subulata*), arborescent (*A. florida*, *A. robusta*), corymbose (*A. divaricata*, *A. hoeksemai*, *A. latistella*), and corymbo-tabulate (*A. valenciennesi*).

DISCUSSION

Highly abundant and diverse *Acropora* populations are uncommon on most inshore reefs in the Gulf of Thailand, making the coral community at Losin Pinnacle a unique potential reservoir of *Acropora* diversity. After the 2010 bleaching event, coral communities in the Gulf of Thailand have been dominated by bleaching-resistant taxa, with a lower coral diversity than in previous periods (Sutthacheep et al., 2013, 2019). Losin Pinnacle has a much higher species diversity, over 75 species, compared to other Thai reefs; 47 species were recorded from twelve sites in the Andaman Sea (Phongsuwan

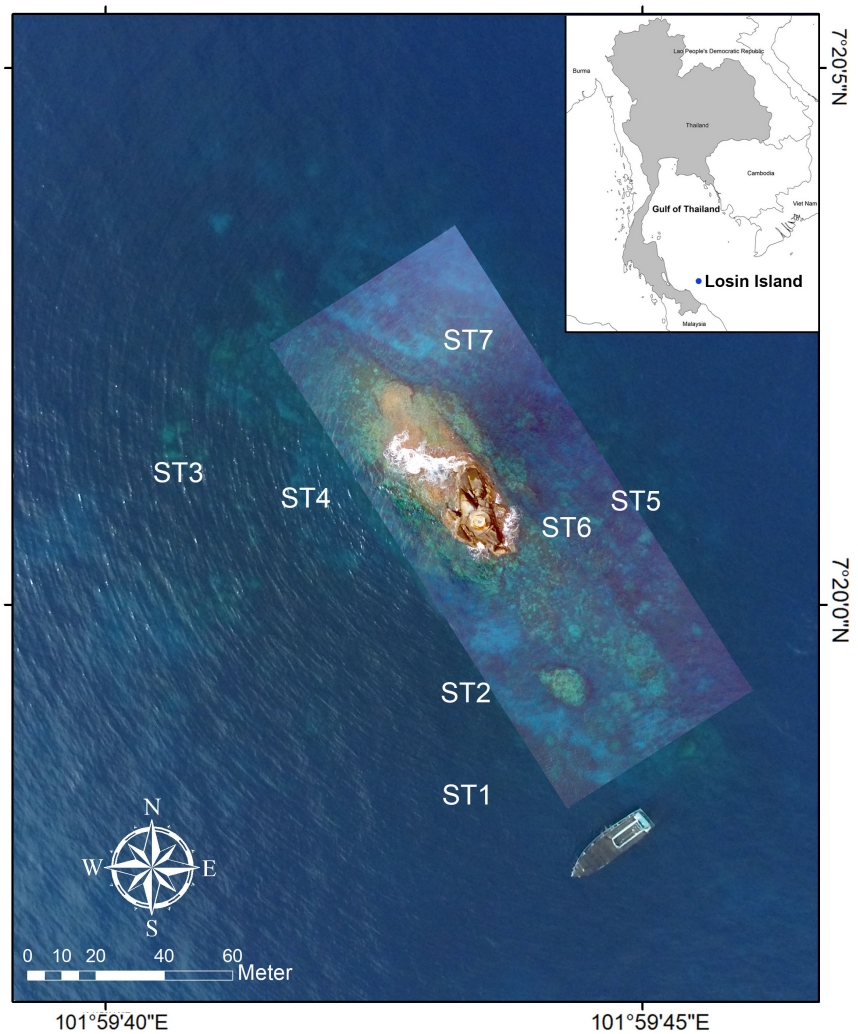


FIGURE 1 | Location of Losin Pinnacle; 7.33360051658N, 101.995261722 E. Shallow survey sites at 10 m comprise ST2, ST4, and ST6, while deep surveys at 20 m comprise sites ST1, ST3, and ST5. All surveys were conducted in an anticlockwise direction, staying parallel to the actual pinnacle. Sites ST1, ST3, ST5, and ST7 were used to collect physical environmental measurements.

and Chansang, 2012), and only 44 species were recorded from six sites in the Gulf of Thailand (Yeemin et al., 2009; Sutthacheep et al., 2012, 2013). Though the overall area of the reef at Losin Pinnacle is relatively small, the coverage of live coral is high relative to other Thai reefs, which typically have live coral cover ranging from approximately 8% to 45% (Phongsuwan et al., 2013; Yeemin et al., 2013a; Sutthacheep et al., 2015). Additionally, Losin reef is rich in fish (111 species, personal communication), and other benthic organisms such as ascidians, octocorals, echinoderms, crustaceans, and sponges (Darumas et al., 2018; Marine and Coastal Resources Research and Development Institute, 2018).

This reef should be considered as a restricted area in which there would be appropriate conservation strategies; specifically, an MPA. The designated MPA would also aim to integrate social and ecological aspects, as previous successful MPAs have done, to increase its effectiveness (Ban et al., 2011). Although it would not be protected against high seawater temperature, setting up an MPA with proper enforcement would reduce the vulnerability of corals to anthropogenic disturbances (Keller et al., 2009; Selig and Bruno, 2010; Wilson et al., 2012) through restricting use of the sea around Losin Pinnacle and limiting excessive use of the reef by tourists or fishermen.

Our baseline coral data, along with awareness and education sessions on coral bleaching, have encouraged the authority (Department of Marine and Coastal Resources) to draft an MPA plan for Losin Reef. After conducting the final public hearing with relevant agencies and stakeholders on October 2018, the proposal submitted to the National Policy and Plan Committee on Marine and Coastal Resources is currently under discussion, and in the future, hopefully, the MPA will be enacted. The next step will be preparation for the legal act, and it is expected to be put into force by the end of 2020. It is strongly suspected that the output from this study will actively support

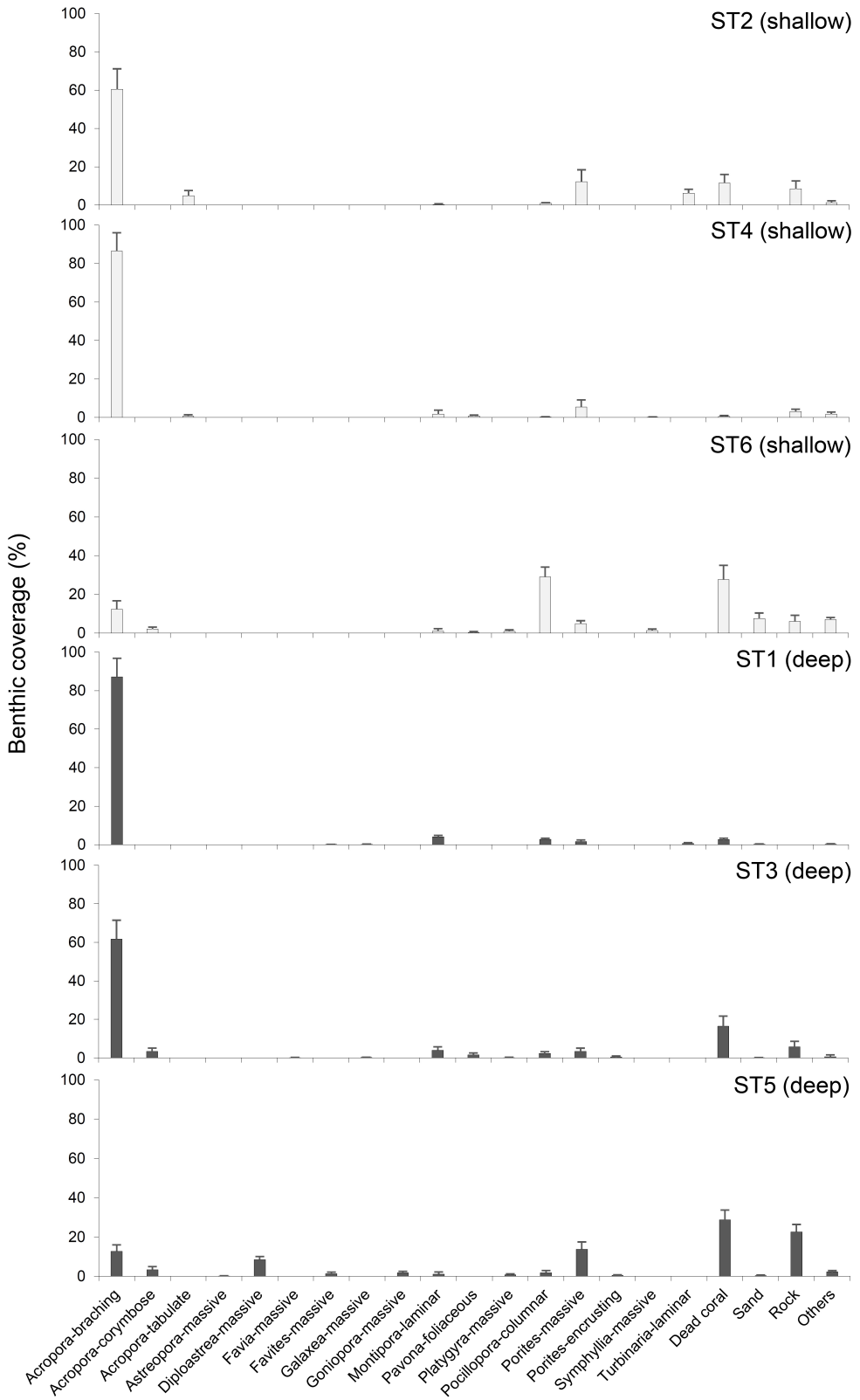


FIGURE 2 | Benthic cover data (mean + SD) observed from three shallow (10 m) and three deep (20 m) stations around Losin Pinnacle during August 2017. Percentage cover was calculated from photographs taken every half meter with three replicates at 30-m belt transects. Corals were divided into growth forms, and several abiotic variables were measured. Others represent benthic organisms other than hard corals.

Thailand's management plan under the Act on the Promotion of Marine and Coastal Resources Management B.E. 2558 (2015). The importance of Losin Pinnacle as a source of coral diversity for other reefs and as a potentially resilient reef to increased ocean temperatures should be further assessed in the future for population genetics and oceanographic studies.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusion of this article will be made available by the authors, without undue reservation, to any qualified researcher.

AUTHOR CONTRIBUTIONS

MY, AC, SN, and UD conceived and planned the main idea. SN supervised the project for surveying the coral reef at Losin Pinnacle and planned the underwater coral studies. NP, MY, and AC contributed to sample collection and the identification of corals. MY, NP, and SN contributed to the data collection and analyzed coral reef status. MY and NP interpreted the results with support from AC. UD, ST, and SP wrote the project on "Draft of the Announcement of the Losin Pinnacle Marine Resources Protected Area" and contributed to public hearings with the support of MY. MY performed the data analysis and wrote the manuscript with support from NP and AC. MY, AC, NP, and SN provided **Supplementary Material**. All of the authors carried out the field studies and discussed the results related to the final manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2020.00130/full#supplementary-material

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