SUCCESSES AT THE INTERFACE OF OCEAN, CLIMATE AND HUMANS

EDITED BY: Nancy Knowlton, Emanuele Di Lorenzo, Fiorenza Micheli and Christopher B. Field

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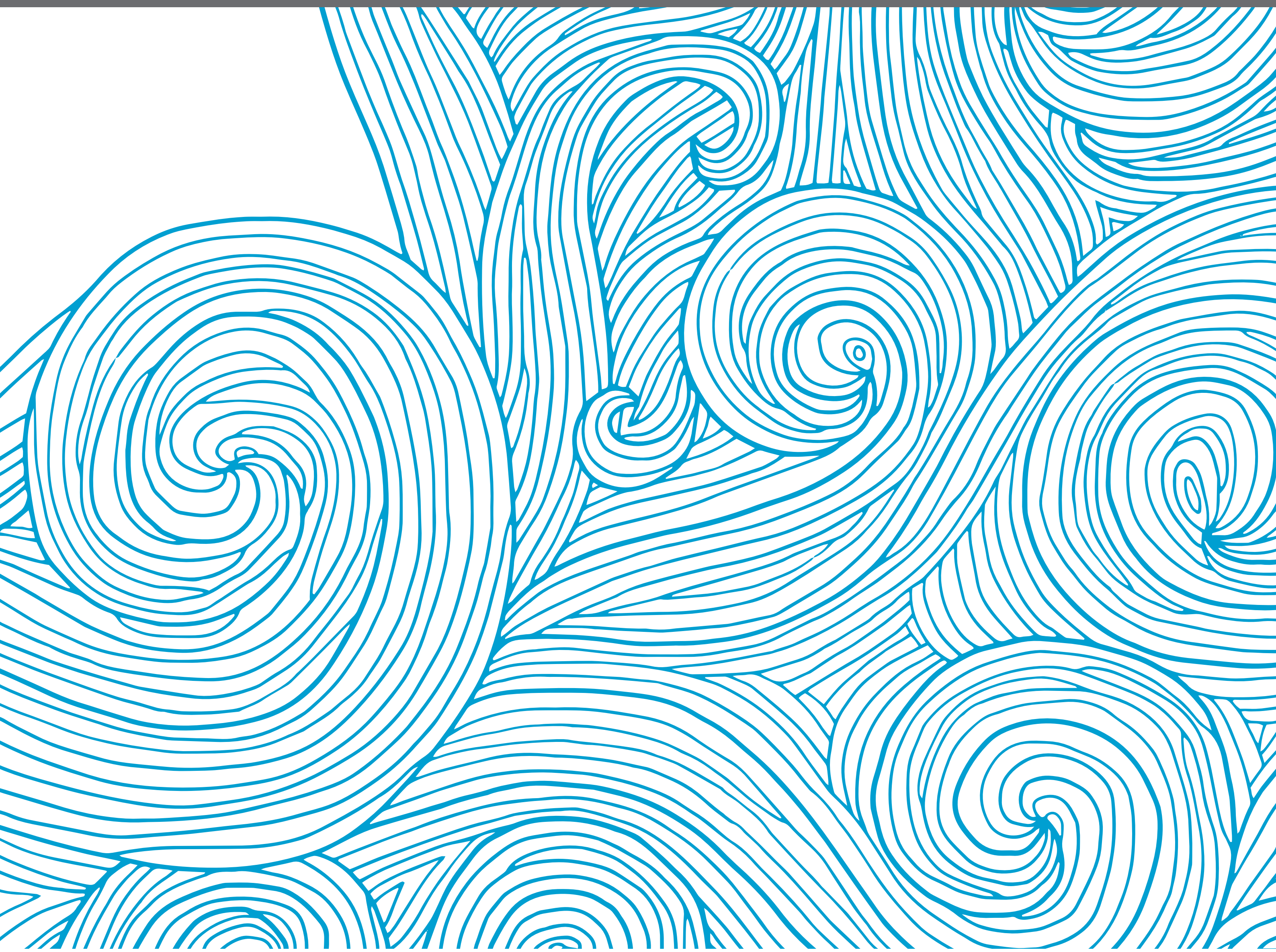

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SUCCESSES AT THE INTERFACE OF OCEAN, CLIMATE AND HUMANS

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ORIGINAL RESEARCH
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Ocean Solutions to Address Climate Change and Its Effects on Marine Ecosystems

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The Paris Agreement target of limiting global surface warming to 1.5–2°C compared to pre-industrial levels by 2100 will still heavily impact the ocean. While ambitious mitigation and adaptation are both needed, the ocean provides major opportunities for action to reduce climate change globally and its impacts on vital ecosystems and ecosystem services. A comprehensive and systematic assessment of 13 global- and local-scale, ocean-based measures was performed to help steer the development and implementation of technologies and actions toward a sustainable outcome. We show that (1) all measures have tradeoffs and multiple criteria must be used for a comprehensive assessment of their potential, (2) greatest benefit is derived by combining global and local solutions, some of which could be implemented or scaled-up immediately, (3) some measures are too uncertain to be recommended yet, (4) political consistency must be achieved through effective cross-scale governance mechanisms, (5) scientific effort must focus on effectiveness, co-benefits, disbenefits, and costs of poorly tested as well as new and emerging measures.

Keywords: climate change, ocean acidification, ocean solutions, global, local, governance

INTRODUCTION

The ocean provides most of the life-supporting environment on the planet. It hosts a large portion of biodiversity, plays a major role in climate regulation, sustains a vibrant economy and contributes to food security worldwide. Severe impacts on key marine ecosystems and ecosystem services are projected in response to the future increase in global mean temperature and concurrent ocean acidification, deoxygenation, and sea-level rise (Hoegh-Guldberg et al., 2014; Pörtner et al., 2014; Gattuso et al., 2015). These impacts scale to CO_2 emissions: they will be considerably worse with a high emissions scenario than with a scenario that limits the temperature increase to 2°C relative to pre-industrial levels (Bopp et al., 2013). Current emission reduction pledges under the 2015 Paris Agreement (UNFCCC, 2015) are, however, insufficient to keep global temperature below +2°C in 2100 relative to pre-industrial level (Rogelj et al., 2016) and to reach targets for the United Nations Sustainable Development Goals. Increased ambition, with additional actions, is therefore required.

Further reductions in atmospheric greenhouse gas emissions are achievable through: (1) a shift from fossil fuels to renewable energy; (2) improved energy efficiency; (3) carbon capture and storage (CCS) at the point of CO_2 generation; and (4) the protection and enhancement of natural carbon sinks (Griscom et al., 2017; Rockström et al., 2017). The risk of failing to meet climate targets via emissions reduction has increased interest in solar radiation management (National Research Council, 2015b) and carbon dioxide removal from the atmosphere (National Research Council, 2015a; Williamson, 2016; Hansen et al., 2017). For example, the implementation of bioenergy with carbon capture and storage is a major component of a roadmap to reduce global emissions from ∼40 Gt CO_2 year^{-1} in 2020 to ∼5 Gt CO_2 year^{-1} by 2050 (Rockström et al., 2017). Such an ambitious roadmap, however, poses significant political, economic, and environmental challenges because of the land, water, and nutrient requirements to produce the biomass (potentially in competition with existing ecosystems, land use, and food production), the cost and feasibility of carbon capture and storage, and the fact that such systems have yet to be proven effective at the required scales (Anderson and Peters, 2016; Smith et al., 2016; Boysen et al., 2017). Additionally, and even under a successful mitigation scenario, impacts are expected at the local scale, hence the need for enhanced adaptation measures.

To date, policy responses to climate change and its impacts have largely focussed on land-based actions (Field and Mach, 2017) while relatively little attention has been paid to ocean-based potential (Rau et al., 2012; Billé et al., 2013), despite the recent launch of the Ocean Pathway initiative by the Presidency of the 23rd Conference of the Parties (COP23) of the United Nations Framework Convention on Climate Change (UNFCCC). The ocean already removes about 25% of anthropogenic CO_2 emissions (Le Quéré et al., 2018) and has the potential to remove and store much more (Rau, 2014). Thus, ocean-based actions could significantly reduce the magnitude and rate of ocean warming, ocean acidification, and sea-level rise, as well as their impacts on marine ecosystems and ecosystem services. They could also play a significant role in helping to reduce global warming and its impacts on the non-ocean surface of the planet – and on human society. However, there may be associated risks to ocean life and people, and there is a lack of guidance for prioritizing ocean-based interventions since there has been relatively little research, development and deployment in this field. Important issues include determining the effectiveness of a given approach in countering changes in climate drivers and/or impacts, possible spatial and temporal scales of deployment, associated positive and negative climate, environmental, economic, and societal impacts (Russell et al., 2012), and hence the implications for ethics, equity, and governance (Preston, 2013; Burns et al., 2016; Williamson and Bodle, 2016).

To fill this gap, we assess the potential of 13 categories of ocean-based measures or schemes to reduce climate-related drivers globally and/or locally (<∼100 km^2), as well as to reduce adverse impacts on selected, important and sensitive marine ecosystems and ecosystem services. The three drivers considered are ocean warming, ocean acidification and sea-level rise, although others such as hypoxia, extreme events, and changes in storminess and precipitation can also be important. We focus on four ecosystems and habitats (warm-water coral reefs, mangroves and salt-marshes, seagrass beds, and Arctic biota) and four ecosystem services (finfish fisheries, fish aquaculture, coastal protection, and bivalve fisheries and aquaculture), which are particularly vulnerable to climate impacts and are critical for livelihoods and food security. The potential of each ocean-based measure is assessed in terms of the following eight environmental, technological, social, and economic criteria: (1) potential effectiveness to increase net carbon uptake and moderate ocean warming, ocean acidification, and sea level rise; (2) technological readiness; (3) lead time until full potential effectiveness; (4) duration of benefits; (5) co-benefits; (6) disbenefits; (7) cost effectiveness; and (8) governability from an international perspective. This expert assessment is based on an extensive literature review and is supported by **Supplementary Materials** (SM) that provide details on the terminology, assessment methods, results, and supporting literature.

CLIMATE-RELATED SENSITIVITY OF OCEAN ECOSYSTEM AND ECOSYSTEM SERVICES

Key Ecosystems Investigated

Ecosystems have different sensitivities to ocean warming, ocean acidification and sea-level rise (**Figure 1A** and section "SM3.3" of the **Supplementary Materials**). Interactions between drivers can be complex: additive, synergistic, or antagonistic (Crain et al., 2008). There are big gaps in multiple-drivers research (Crain et al., 2008; Riebesell and Gattuso, 2015) but experimental

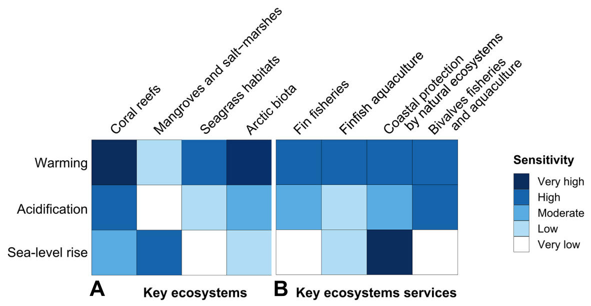

FIGURE 1 | Sensitivity of key ecosystems **(A)** and ecosystem services **(B)** to key climate-related drivers. The scores of this expert assessment and their justification are provided in section "SM3.3" of the **Supplementary Materials**.

strategies to assess the biological ramifications of multiple drivers of global ocean change have become available (Boyd et al., 2018).

Of the four ecosystems or habitats considered here, coral reefs and Arctic biota are the most imminently threatened and will be affected to a greater degree sooner than others, with high risk that key functions will be lost globally, as identified in the 5th assessment report (AR5) of the Intergovernmental Panel on Climate Change (IPCC) (Hoegh-Guldberg et al., 2014; Pörtner et al., 2014). Coral reefs are very sensitive to ocean warming and acidification (Hoegh-Guldberg et al., 2007; Gattuso et al., 2014; Hughes et al., 2017a). They have suffered extensive losses in the past three decades due to high sea surface temperature combined with local stressors such as overfishing, destructive fishing, coastal development, and pollution. All projections indicate that the thermal conditions driving major losses will increase in frequency and exceed thresholds for the majority of reefs by 2050 (Gattuso et al., 2014; Pörtner et al., 2014). Over the 21st century under the high emissions Representative Concentration Pathway RCP8.5 scenario (van Vuuren et al., 2011), 99% of the world's coral reefs are expected to experience annual severe bleaching due to thermal stress (van Hooidonk et al., 2016). The thermal sensitivity of coral reefs is compounded by ocean acidification (Hoegh-Guldberg et al., 2014), which diminishes coral growth and calcification (Albright et al., 2018) and can lead to increased bioerosion and vulnerability to storm damage (Andersson and Gledhill, 2013).

Arctic biota are also highly sensitive to climate change, particularly ice-associated biota that are rapidly declining in Arctic summers (Wassmann et al., 2010; Pörtner et al., 2014; Kohlbach et al., 2017). Within the Arctic, ecosystem responses vary greatly depending on ambient variability, degree of warming, and nutrient advection (Hunt et al., 2016). Warming and freshening may also impact ecosystem production by differentially increasing respiration rates and reducing nutrient supply (Duarte et al., 2012) as well as enhancing the degree of ocean acidification due to freshening (Pörtner et al., 2014). Arctic organisms that seem particularly sensitive to ocean acidification include calcifiers such as bivalves and planktonic pteropods that are key links in ocean food webs (Comeau et al., 2010; Duarte et al., 2012).

Mangroves and saltmarshes are highly sensitive to sea-level rise (Kirwan and Megonigal, 2013; Lovelock et al., 2015), particularly where coastal development and steep topography block landward migration and insufficient sediment is delivered to support accretion. A preliminary global modeling effort suggests that a 50 cm sea-level rise by 2100 would result in a loss of 46 to 59% of global coastal wetlands (up to 78% loss under 110 cm rise), but losses are sensitive to assumptions about human coastal development and may be reduced if additional tidal hydrodynamic feedbacks are included (Spencer et al., 2016). Warming and acidification are not projected to have significant direct effects on mangroves and saltmarshes, but may have positive or negative effects at local scales due to changes in species composition, phenology, productivity, and latitudinal range of distribution (Ward et al., 2016).

Temperate seagrass ecosystems are sensitive to ocean warming. For example, the thermal regime of the Mediterranean Sea already exceeds the upper thermal limit of the endemic *Posidonia oceanica* in some areas (Marbà and Duarte, 2010; Jordà et al., 2012). Seagrass and fleshy algae may expand in Arctic regions with warming and loss of ice cover (Krause-Jensen and Duarte, 2014). Some may benefit from carbonate chemistry changes associated with ocean acidification as their photosynthesis is CO_2-limited (Raven and Beardall, 2014) but sensitive calcifiers growing in the meadows are negatively impacted (Martin et al., 2008).

Key Ecosystem Services Investigated

The ecosystem services considered in this study are all highly sensitive to ocean warming (Weatherdon et al., 2016; **Figure 1B** and section "SM3.3" of the **Supplementary Materials**). Global potential fisheries catches and species turnover, for instance, are projected to decrease by about 3 Mt and increase by 10%, respectively, for every 1°C of global surface warming (Cheung et al., 2016). These patterns are similar for finfish and shellfish aquaculture, as ~90% of current finfish and shellfish mariculture production is from open-water farming where environmental conditions closely match those in the nearby ocean (Callaway et al., 2012). Shellfish fisheries and mariculture, in particular, are threatened by the combined effects of warming (Mackenzie et al., 2014), ocean acidification (Barton et al., 2012; Gazeau et al., 2013) and deoxygenation (Gobler et al., 2014). Despite possible genetic adaptation over generations (Thomsen et al., 2017), impacts on shellfish are expected to be high to very high when CO_2 concentrations exceed those projected for 2100 in the low to moderate RCP2.6 and 4.5 CO_2 emissions scenarios (Gattuso et al., 2015; Cooley et al., 2016). In addition, finfish mariculture often focuses on high trophic level species that are dependent on wild capture fisheries for feed (Troell et al., 2014) and some operations still largely rely on wild captured fish fry and juveniles (Diana, 2009). Thus, mariculture is likely to be subject to similar climatic stresses as fish stocks in the wild.

The sensitivity of coastal protection, notably wave attenuation and shoreline stabilization, to climate-related drivers differs for each ecosystem considered (Spalding et al., 2014). The cumulative impacts of increasing sea-surface temperature, ocean acidification, and non-climatic stressors such as land-based pollution reduce reefs' ability to keep pace with sea-level rise (Yates et al., 2017). The consequences of sea-level rise on biologically structured coastal ecosystems raise concerns as these habitats are estimated to currently reduce wave height by 30 to 90% (in order of highest to lowest wave reduction: coral reefs, saltmarshes, mangroves, and seagrasses) (Fonseca and Cahalan, 1992; Duarte et al., 2013; Ferrario et al., 2014; Narayan et al., 2016). Historical global losses in coastal ecosystems [30 to 50% for mangroves since the 1940s, 29% for seagrass since 1879, 25% for saltmarshes since the 1800s (Waycott et al., 2009; Mcleod et al., 2011)] and degradation of coral reefs [30–75% since prehuman times (Pandolfi et al., 2003)] have already reduced their potential to provide ecosystem services. Projections suggest that 90% of coral reefs worldwide could be lost if warming exceeds 1.5°C (Frieler et al., 2013).

OCEAN-BASED SOLUTIONS

Four types of actions to reduce the scale and impacts of climate change are considered (**Figure 2**): (1) reduction of atmospheric greenhouse gas concentrations, (2) solar radiation management, (3) protection of biota and ecosystems, and (4) manipulation of biological and ecological adaptation. The actions in the first two categories (referred to as global actions hereafter, although some forms of solar radiation management could be local) aim to either reduce the main cause of climate change at the global scale (primarily the increase in atmospheric CO_2 concentration) or to counteract warming through increasing albedo in the atmosphere or at the Earth's surface, thereby increasing the proportion of solar radiation that is reflected back to space. The actions in the other two categories (referred to as local actions hereafter) aim to reduce the risk of climate change impacts locally, either by reducing the locally experienced drivers (site-specific acidification and warming, and relative sea-level rise) and/or reducing the sensitivity of organisms and ecosystems to these drivers (Bates et al., 2017; Cheung et al., 2017). *Vegetation* and *alkalinization* (see **Box 1** and section "SM1" of the **Supplementary Materials** for descriptions) are evaluated for both global and local aims as they can be deployed globally to reduce changes in climate-related drivers and impacts, as well as locally to reduce the sensitivity of marine ecosystems and services to specific drivers such as relative sea-level rise and ocean acidification.

Other ocean-based measures have been proposed but little research has been conducted on their potential. They include large-scale seaweed aquaculture for supplementing cattle feed to reduce methane emissions and counteract acidification locally (Machado et al., 2016; Duarte, 2017). Abiotic methods of removing or stripping CO_2 from seawater have also been proposed or demonstrated in the laboratory (Eisaman et al., 2012; Willauer et al., 2014; Koweek et al., 2016), as well as marine-based interventions that increase uptake and reduce emissions of other greenhouse gases such as CH_4 and N_2O (e.g., Poffenbarger et al., 2011; Stolaroff et al., 2012). Research and testing of new, unconventional methods of ocean and climate management are in their infancy, and additional methods are likely to emerge.

Whereas some of the solutions assessed here are still at a very-early or experimental stage, others have been implemented and refined over many decades, though not always specifically designed to address climate change impacts. The global implementation of *renewable energy*, *vegetation*, *eliminating overexploitation,* and *protection* exhibit a sharp acceleration in the past two decades (**Figure 3**). For example, global cumulative offshore wind potential has grown 3-fold in less than 5 years to reach 15,000 MW in 2016 (Global Wind Energy Council, 2016). MPAs now cover more than 3% of the global ocean (Boonzaier and Pauly, 2016), 7% of the overexploited fish stocks have been rebuilt (Kleisner et al., 2013) and the global area of avoided loss of mangroves has been estimated at 40,000 km^2 (Hamilton and Casey, 2016).

POTENTIAL TO REDUCE KEY OCEAN DRIVERS

Effectiveness of the Measures and Duration of Their Effects

To estimate effectiveness, we first assess the potential of each measure –assumed here to be implemented at its maximum physical capacity– to bridge the gap between the high-emissions trajectory (RCP8.5, our baseline scenario) and a stringent emission-reduction scenario (RCP2.6) expected to keep mean global temperature increase below 2°C by 2100 (van Vuuren et al., 2011) (see section "SM2" of the **Supplementary Materials**). The differences between RCP8.5 and RCP2.6 in the year 2100 are estimated to be $\sim$1,400 Pg C for avoided emissions; $\sim$2°C for reduced sea surface warming; $\sim$0.25 pH units for avoided sea surface acidification; and a reduction in sea-level rise of between 0.26 and 1.1 m (Jones et al., 2013; DeConto and Pollard, 2016).

The effectiveness of the global measures is assessed in terms of maximum possible effectiveness to reduce ocean warming, ocean acidification, and sea-level rise (**Figure 4A**), and duration of the effect (**Figure 4B**). This maximum effectiveness is theoretical and almost certainly not achievable but provides the full potential of each approach. Two of the global solutions, *renewable energy* and *alkalinization*, stand out as having the highest theoretical potential for addressing all drivers (**Figure 4A**). This is obvious for *renewable energy* because of the enormous energy potential of tides, waves, ocean currents, and thermal stratification, estimated at up to 7,400 EJ year^{-1} (Rogner et al., 2000; Lewis et al., 2011) and well exceeding future human energy needs. Any replacement of fossil fuels by marine renewables results in permanently avoided greenhouse gas emissions.

A similarly large and permanent intervention could be provided by large-scale *alkalinization*, by which CO_2 is consumed

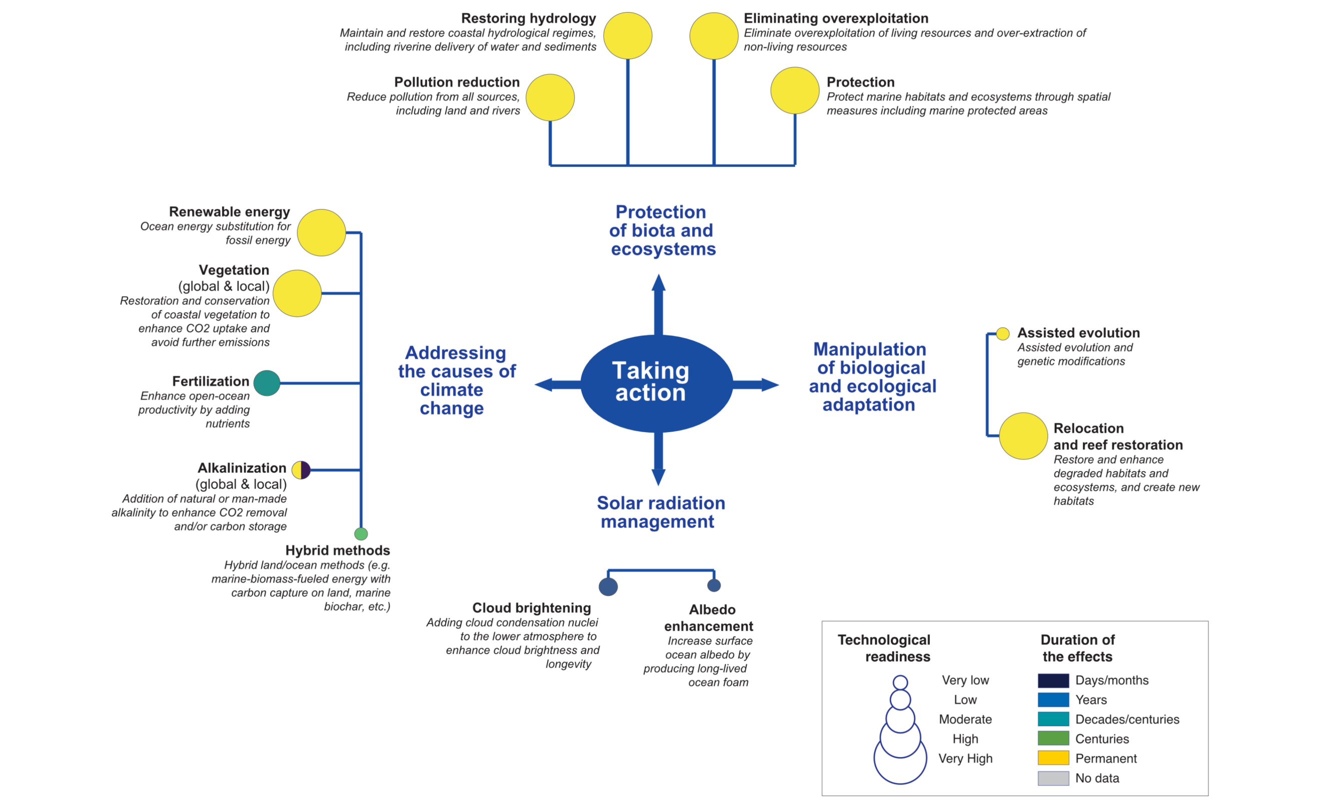

FIGURE 2 | Potential ocean solutions. Four main groups are considered: addressing the causes of climate change (i.e., reducing anthropogenic greenhouse gas emissions or increasing the long-term removal of greenhouse gases, primarily CO_2), solar radiation management, protection of biota and ecosystems (habitats, species, resources, etc.), and manipulation of biological and ecological adaptation.

and stored either as dissolved bicarbonate and carbonate ions or as precipitated calcium carbonate, neutralizing ocean acidity. However, the feasibility and benefits of doing this must be weighed against the financial costs and environmental impacts of mining or producing vast quantities of alkaline material, distributed at global scales, and the potential biotic impacts of the trace elements or contaminants that alkalinity might contain (Renforth and Henderson, 2017).

Land-ocean *hybrid methods* greatly expand the mitigation potential offered by either land-based or ocean-based approaches individually. For example, the use of marine biomass for bioenergy with carbon capture and storage (BECCS) fuel eliminates limitations on terrestrial fuel capacity posed by competition for land, water, and nutrients. In turn, conversion of CO_2 from land-based biomass energy to ocean alkalinity and subsequent storage in the ocean greatly expands CO_2 storage capacity and beneficial use (via countering ocean acidification) relative to more conventional CCS approaches. However, a comprehensive understanding of the full range of options, and their costs, benefits and tradeoffs requires further research (Rau, 2014).

Albedo enhancement also has a very large potential effectiveness in moderating warming (**Figure 4A**), as a relatively small enhancement of the albedo of the dark ocean surface by less than 0.05 could compensate the entire GHG-driven perturbation in the Earth's radiation balance (Crook et al., 2016; Garciadiego Ortega and Evans, 2018). However, the duration of the effect is only as long as the albedo stays high, likely to be days to months for ocean foams (**Figure 4B**) and, as SRM in general, it does not limit ocean acidification as atmospheric CO_2 concentration remains elevated (Tjiputra et al., 2016). Similar considerations apply to marine *cloud brightening*, although modeling studies indicate more limited effectiveness (Kravitz et al., 2013; Stjern et al., 2017).

Other potential solutions face physical and/or biogeochemical limitations (**Figure 4A**). A global deployment of iron *fertilization* for 100 years could sequester a maximum of ~70 Pg C (ref. Aumont and Bopp, 2006) because other nutrient or light limitations occur when marine algae are iron-replete (Oschlies et al., 2010). Some measures demonstrate limited potential for reducing warming, acidification and sea-level rise at global scales, such as *vegetation* for instance. Even with very high carbon storage and avoided net emissions, the *vegetation* measure is constrained by the limited global area of potentially vegetated habitats, although with some scope to artificially expand that area; e.g., via seaweed aquaculture (Duarte et al., 2017; Hawken, 2017).

Local measures have a relatively low effectiveness to reduce warming, acidification, and sea-level rise at the global scale (**Figure 4A**). However, some have a high to very high effectiveness

BOX 1 | Ocean-based solutions. Measures that address the causes of global climate change either reduce anthropogenic greenhouse gas emissions or increase their long-term removal from the atmosphere. Five measures are considered in this group, including negative emissions technologies (see Minx et al., 2018) which are critical for achieving the long-term climate goals of the Paris Agreement (UNFCCC, 2015). (1) Ocean-based renewable energy (hereafter ***renewable energy***) comprises the production of energy using offshore wind turbines and harvesting of energy from tides, waves, ocean currents, and thermal stratification (Pelc and Fujita, 2002). (2) The restoration and conservation of coastal vegetation (hereafter ***vegetation***), primarily saltmarshes, mangroves and seagrasses (also referred to as "blue carbon ecosystems"), seeks to enhance their carbon sink capacity and avoid emissions from their existing large carbon stocks if degraded or destroyed (Mcleod et al., 2011; Herr and Landis, 2016). This measure is considered not only in terms of global implementation – i.e., assuming theoretical worldwide conservation and restoration of all such habitats that have been degraded or lost due to human activities – but also local implementation, providing local mitigation and adaptation benefits in addition to other co-benefits. (3) ***Fertilization*** involves the artificial increase in the ocean's primary production and, hence, carbon uptake by phytoplankton in the open ocean, to be achieved primarily by adding soluble iron to surface waters where it is currently lacking, mostly in mid-ocean gyres and the Southern Ocean (Aumont and Bopp, 2006). (4) ***Alkalinization*** describes the addition of a variety of alkaline substances that consume CO_2 and/or neutralize acidity (Rau, 2011; Renforth and Henderson, 2017), primarily achieved by raising the concentration of carbonate or hydroxide ions in surface waters, and thereby shifting the associated chemical equilibria in seawater to increase the oceanic uptake of atmospheric CO_2. The feasibility and effectiveness of adding alkalinity are considered at both global and local scales. In either case the alkalinity would be derived from land-based mineral or synthetic chemical sources or from locally available marine material (e.g., waste shells). The alkalinity would then require transport to and distribution within the marine environment. (5) Land-ocean ***hybrid methods*** include the use of the ocean and its sediments to store biomass, CO_2 or alkalinity derived from terrestrial sources. Examples are crop residue storage on the seafloor (Strand and Benford, 2009), marine storage of CO_2 from land-based bio-energy or from direct air capture of CO_2 (Sanz-Pérez et al., 2016) and conversion of such CO_2 to alkaline forms for ocean storage (Rau, 2011). *Hybrid methods* also include techniques involving marine-to-land transfers, such as using marine biomass to fuel biomass energy with carbon capture and storage (BECCS) on land or using such biomass to form biochar as a soil amendment.

Another area of action to counter global and ocean warming (but which does not directly address the greenhouse gas cause) is solar radiation management (SRM, also known as sunlight reflection methods). Several schemes were described, including stratospheric aerosol injection (National Research Council, 2015b). Two ocean-based schemes are considered here. (6) Marine cloud brightening (hereafter ***cloud brightening***) involves the large-scale aerial spraying of seawater or other substances into the lower atmosphere to increase the amount of sunlight clouds reflect back into space (Latham et al., 2012; Kravitz et al., 2013). Sub-global implementation could also be considered (Latham et al., 2013). (7) Increased surface ocean albedo (hereafter ***albedo enhancement***) is here considered to be achieved by long-lived ocean micro-bubbles or foams, produced either by commercial shipping (Crook et al., 2016) or by vessels dedicated to that task.

Four measures relate to the protection of biota and ecosystems. (8) ***Reducing pollution*** refers to decreasing release of anthropogenic, harmful substances. Pollution can exacerbate hypoxia and ocean acidification especially in coastal waters (Cai et al., 2011) while increasing the sensitivity of marine organisms and ecosystems to climate-related drivers (Alava et al., 2017). (9) Restoring hydrological regimes (***restoring hydrology***) relates to the maintenance and restoration of marine hydrological conditions, primarily in coastal waters, including both the tidal and riverine delivery of water and sediments, to alleviate local changes in climate-related drivers (Howard et al., 2017). (10) ***Eliminating overexploitation*** includes ensuring the harvest and extraction of living resources are within biologically safe limits for sustainable use by humans and to maintain ecosystem function and, in the case of non-living resources (e.g., sand and minerals), in levels that avoid irreversible ecological impacts. For example, in over-exploited ecosystems, pelagic species that are smaller and faster turnover generally increase in dominance (Cheung et al., 2007). Abundance of these pelagic species tends to track environmental conditions more closely than large demersal fishes (Winemiller, 2005), the latter are often depleted in over-exploited systems (Cheung et al., 2007). Thus, fisheries with increased dominance of pelagic species are generally more sensitive to changes in environmental conditions from climate change (Planque et al., 2010). Although species with higher turnover rates may theoretically have more capacity to adapt evolutionarily to environmental changes (Jones and Cheung, 2018), the scope and rate of such adaptive response for most fishes are unclear (Munday et al., 2013). Also, over-exploited fish stocks with largely reduced abundance may also have reduced genetic diversity and variability, and consequently the population will have a reduced scope for adaptation under climate change. (11) The protection of habitats and ecosystems (***protection***) refers to the conservation of habitats and ecosystems, primarily through marine protected areas (MPAs). For example, increased abundance of marine species is expected to enhance productivity of the surrounding areas which can help buffer against climate impacts and increase resilience (Roberts et al., 2017).

In the category "manipulation of biological and ecological adaptation" of organisms and ecosystems to the changing ocean conditions, two measures are assessed. (12) ***Assisted evolution*** involves large-scale genetic modification, captive breeding and release of organisms with enhanced stress tolerance (van Oppen et al., 2015). (13) ***Relocation and reef restoration*** involves not only the restoration of degraded coral and oyster reefs (e.g., van Oppen et al., 2017), but also their enhancement and active relocation, with the potential creation of new habitats and use of more resilient species or strains. Note that restoration and protection of vegetated coastal habitats (seagrasses, mangroves, and saltmarshes) is considered in the ***vegetation*** measure.

to moderate local ocean acidification (*pollution reduction* and *alkalinization*) and relative sea-level rise (*vegetation*, *protection*, *restoring hydrology*, as well as *relocation and reef restoration*).

The duration of the effects varies greatly between the different methods (**Figure 4B**). It is close to permanent for *renewables* as long as the infrastructure is maintained. The effects of protection are also considered permanent as long as MPAs are enforced, although future climate change will decrease their ability to provide climate mitigation and adaptation benefits (Bruno et al., 2018). The effects of *vegetation* can be close to permanent as long as these ecosystems are maintained or increased in the face of natural and anthropogenic pressures. In contrast, the effects of *fertilization* have a finite duration. Once iron fertilization is stopped, a large portion of the additional ocean carbon uptake will outgas back to the atmosphere on decadal to centennial time scales (Aumont and Bopp, 2006). By capturing and storing CO_2 for long time periods or permanently, *alkalinization* and *hybrids methods* such as conversion of CO_2 to ocean alkalinity or marine BECCS generally have long duration of the effect. In contrast, the effect of *albedo enhancement* and *cloud brightening* is short-lived (days to weeks). The loss of most benefits following abrupt termination is a characteristic of all SRM schemes (Jones et al., 2013). It is projected to increase both ocean and land temperature velocities to unprecedented speeds (Trisos et al., 2018).

Technical Feasibility and Cost Effectiveness

Technical feasibility is evaluated by considering current technological readiness (ranging from schemes at the concept stage to schemes already deployed) and for lead time until full potential effectiveness, i.e., the time needed to reach full implementation (ranging from days to decades; see section

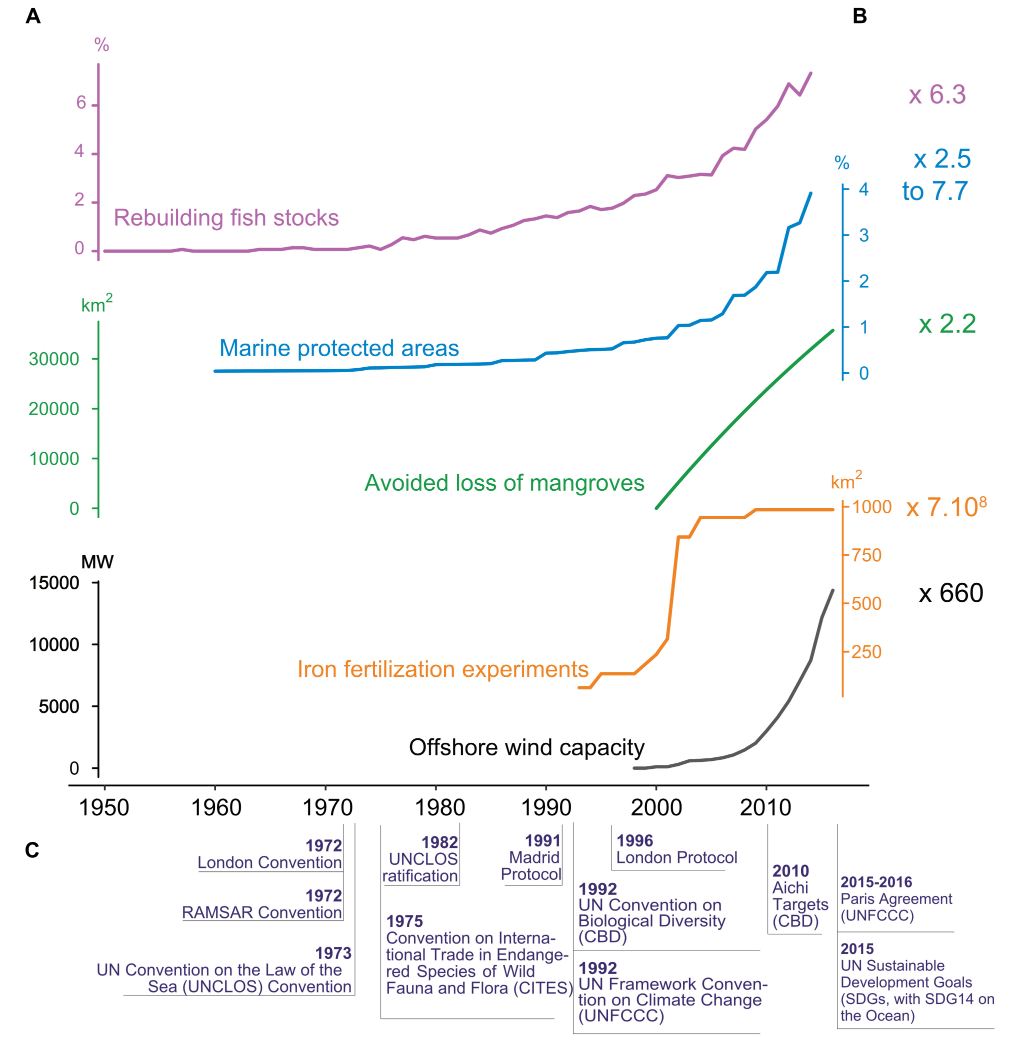

FIGURE 3 | Contemporary history of the global implementation of some ocean solutions. **(A)** Recent changes in the global cumulative offshore wind potential (European Wind Energy Association, 2011; Global Wind Energy Council, 2016), global cumulative surface of ocean iron fertilization experiment patches (Yoon et al., 2016), global area of avoided loss of mangroves (Valiela et al., 2001; Hamilton and Casey, 2016), rebuilding of fish stocks (Kleisner et al., 2013) (in % of total fish stocks), and global cumulative surface of MPAs (Boonzaier and Pauly, 2016) (in % of the global ocean surface). **(B)** Future progress needed to reach full implementation of targets for all measures above, i.e., 300 EJ year^{-1} for offshore wind, all ocean high nutrient and low chlorophyll areas for iron fertilization, 10 and 30% of the global ocean for MPAs (Convention for Biological Diversity, 2010; O'Leary et al., 2017), all overexploited and collapsed fish stocks in the process of rebuilding (in 2014, 46% of the total fish stock was overexploited or collapsed) (Cheung et al., 2017), and pre-disturbance extent of mangroves (Valiela et al., 2001). **(C)** Launch date of some major international conventions or protocols providing governance frameworks for the solutions described in panel **(A)**. For example, initially dealing with marine pollution, the London Convention and the London Protocol also prohibit ocean iron fertilization (except for research purposes).

"SM2" of the **Supplementary Materials**). Two local measures have the highest technical feasibility (**Figure 4B**): *protection* and *restoring hydrology*. *Vegetation* (both global and local) and *renewable energy* also have a high technical feasibility, closely followed by *eliminating overexploitation*, *reducing pollution* and *relocation and reef restoration*. Five global schemes have the lowest technical feasibility: *fertilization*, *cloud brightening*, *alkalinization*, *albedo enhancement,* and *hybrid methods*. The local measure *assisted evolution* also scores very low on this criterion. These low scores generally reflect lack of testing and deployment at scale, thus they also possess high uncertainty.

The cost effectiveness of the global and local solutions is assessed, in US$ per tonne of CO_2 emissions reduced and in US$ per hectare of surface area of implementation, respectively (**Figure 4E** and section "SM3.5" of the **Supplementary Materials**). The costs considered here are best estimates from the literature for the direct monetary costs of implementation. The non-monetary costs of implementation are considered through assessing co-benefits, disbenefits, and governability, as discussed below. Since cost effectiveness is a relative metric, it does not reflect the effectiveness of a measure to reduce changes in the drivers. For instance, *cloud brightening* is cost-effective despite having a moderate maximum effectiveness to

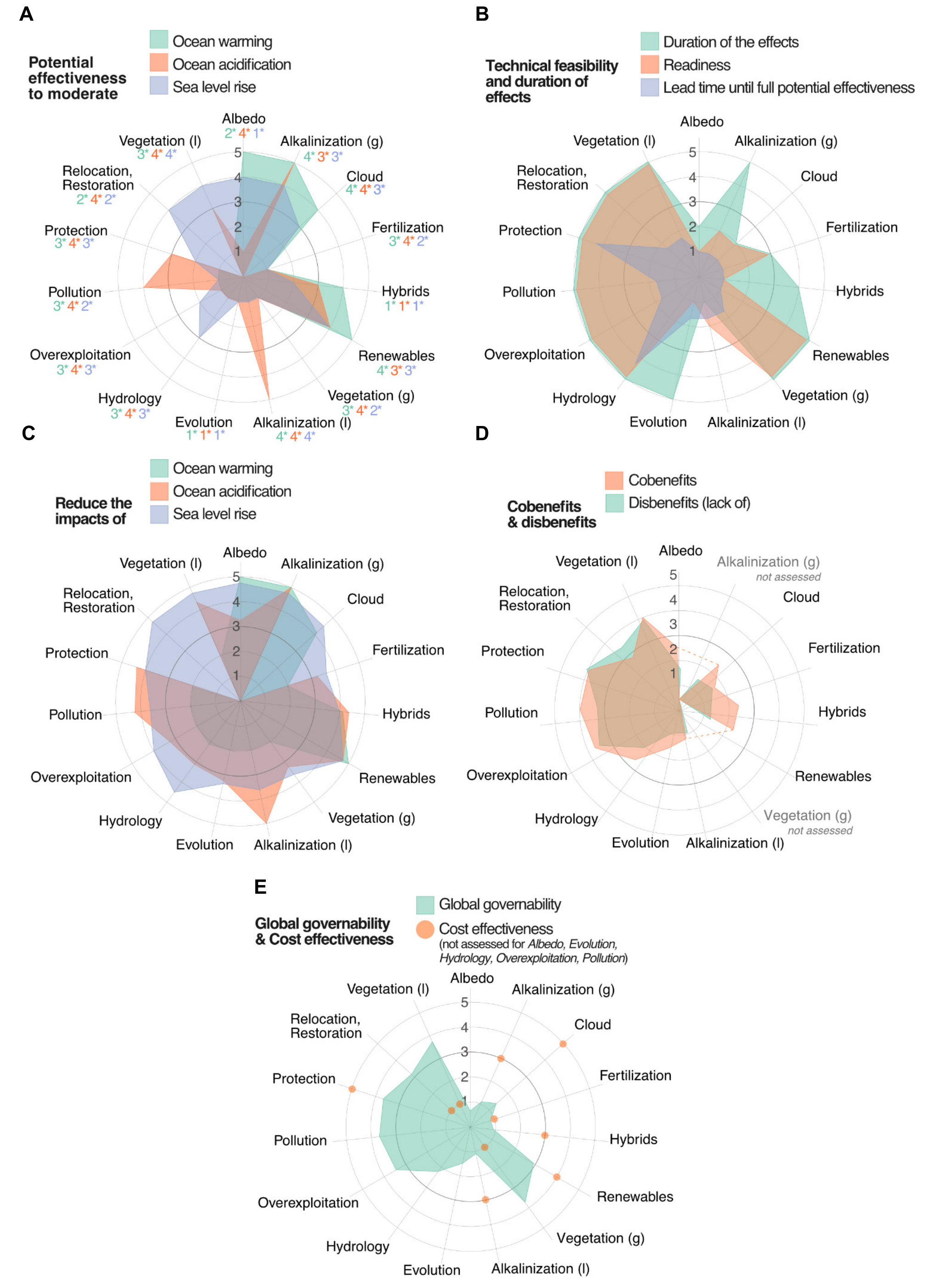

FIGURE 4 | Assessment of ocean-based measures to address key ocean drivers. Scores 1 to 5: very low, low, moderate (thicker circle), high, and very high. Confidence levels of the potential effectiveness to moderate ocean warming, ocean acidification, and sea-level rise are shown in panel **(A)** (1* to 5*; very low, low, moderate, high, very high; see section "SM2.1" of the **Supplementary Materials**). Details on the assessment can be found in section "SM3" of the **Supplementary Materials**.

moderate ocean warming, ocean acidification, and sea-level rise (**Figures 2, 4A**). Restoration of *vegetation* to increase CO_2 capture has a very low cost effectiveness but conservation of vegetation to avoid further emissions is very cost-effective. For example, conserving mangroves to avoid further CO_2 emissions is considerably cheaper than restoring mangroves to enhance CO_2 uptake [4–10 vs. 240 US$/t CO_2 (Siikamaki et al., 2012; Bayraktarov et al., 2016)]. *Cloud brightening, protection,* and *renewable energy* have the highest cost efficiency while *albedo enhancement*, *vegetation* and *relocation and reef restoration* have the lowest. Note that cost effectiveness generally increases over time and with increasing scale of implementation, due to learning and economies of scale, and that there is uncertainty in many of these estimates (see section "SM3.5" of the **Supplementary Materials**) as reflected in the low levels of confidence. This generally is a consequence of lack of economic data from testing/deployment of many of these methods at relevant scales.

Global Governability

Governance is the "*effort to craft order, thereby to mitigate conflict, and realize mutual gains*" (Williamson, 2000) amongst actors from public, private, and civil society sectors. Here, we assess the governability of global and local ocean measures in terms of the potential capability of the international community to implement them, managing associated conflicts and harnessing mutual benefits (see section "SM2.9" of the **Supplementary Materials**). We focus on the international dimension of decision and action to reflect the global scope of the study, despite the fact that we recognize that global and local measures do not face the same constrains for implementation – e.g., bi- or multi-lateral diplomatic issues for the former (e.g., Smit, 2014; Cinner et al., 2016; Rabitz, 2016) and local institutional and population reluctance challenges for the latter (e.g., Cinner et al., 2016).

On that basis, the governability of a scheme increases with its effectiveness (Ostrom, 2007), the predictability of its effects (Hagedorn, 2008; Ostrom, 2009), its co-benefits, the absence of disbenefits together with the presence of national-level net benefits, the presence of enabling institutions and the absence of constraining institutions, and higher normative consensus amongst relevant actors (Abbott and Snidal, 1998; Barrett, 2005). Global governability is likely to be much higher when there are national-level net benefits (i.e., national benefits outweigh the negative environmental impacts and national costs of implementation), since single nation states may then implement measures without having to rely on international cooperation (Kaul et al., 1999). This is the case for *protection*, *vegetation* as well as for *relocation and reef restoration* (**Figure 4E** and section "SM3.6" of the **Supplementary Materials**). Conversely, ocean-based SRM measures (*cloud brightening* and *albedo enhancement*), while being more effective in addressing drivers globally, are considered to have low governability because their implementation generally involves international cooperation to solve the free-riding dilemma with regard to global public goods (Pasztor et al., 2017). Thus nations are likely to be reluctant to unilaterally take on extra costs that may reduce their own economic competitiveness (Preston, 2013; Rabitz, 2016; Williamson and Bodle, 2016). Additionally, SRM measures entail potentially significant disbenefits and high uncertainties (**Figures 4D**, **5**; sections "SM3.4 and SM3.4.3" of the **Supplementary Materials**), which further reduce their present governability. *Renewable energy* is in an intermediate position: renewables are becoming more economically competitive compared with fossil-fuel based energy, thereby providing national-level incentives to implementation. Taken together, the scores exhibit a fundamental tradeoff in climate policy: global measures are more effective than local ones in addressing the climate problem, but they are in general more difficult to implement due to challenges in global governance.

POTENTIAL TO REDUCE IMPACTS ON ECOSYSTEMS

Reducing the climate-related impacts depends on two attributes, the effectiveness to reduce exposure to warming, acidification, and sea-level rise (**Figure 4A**; sections "SM3.1 and SM3.2" of the **Supplementary Materials**) and the sensitivity of ecosystems to changes in these drivers (**Figure 1**; section "SM3.3"). Differences in these attributes lead to different reduction of impacts both among drivers and ecosystems (**Figure 5**). For example, *renewable energy* consistently scores very high in its combined effectiveness to reduce the impacts because it reduces exposure to all three drivers. In contrast, *relocation and reef restoration* is one of the less effective measures in reducing impacts because, despite the fact that restoration can reduce relative sea-level rise, it does not necessarily reduce exposure to ocean warming and acidification *in situ* unless the relocation involves species or habitat transfers to localities that are cooler and/or have higher pH. Another example is *albedo enhancement*, the effectiveness of which is very high to reduce the impacts of warming, high for sea-level rise and very low for acidification. Thus aside from solutions like massive and rapid deployment of marine renewable energy, multiple and in some cases non-traditional solutions targeting different drivers may be needed, the combination of which will be ecosystem-specific. For example, solutions that target warming and acidification are more important to reduce the impacts on coral reefs and Arctic biota, whereas solutions that are most effective to reduce the impacts of sea-level rise will be more relevant for mangroves and saltmarshes.

While the most effective measures to reduce exposure to all three drivers are the global ones (**Figure 4A**), they do not generally reduce the sensitivity of the ecosystems to climate-related drivers. In contrast, local solutions have low or moderate effectiveness to reduce changes in climate-related drivers. They aim to moderate impacts primarily through reducing non-climatic drivers that affect the health and resilience of coastal ecosystems and marine environments such as pollution, overexploitation, overfishing, and coastal development (Halpern et al., 2015). Thus, local solutions have a high level of co-benefits and generally induce a low level of disbenefits since many have a long history of successfully mitigating non-climate stressors – the value of which is considered in this study as co-benefits (**Figure 5**). The most effective measures across all ecosystems (high to very high effectivenesses to reduce the impacts of ocean warming, ocean acidification, and sea-level rise; **Figure 4C**) are *renewable energy*, *alkalinization*, *hybrid methods*, *vegetation* (local) and *albedo enhancement*, with *renewable energy* showing the greatest combined effectiveness. *Protection*, *restoring hydrology*, and *eliminating overexploitation* also score relatively high to reduce impacts on seagrass habitats, mangroves and saltmarshes (**Figure 5**). *Relocation and reef restoration* and *cloud brightening* consistently have the lowest

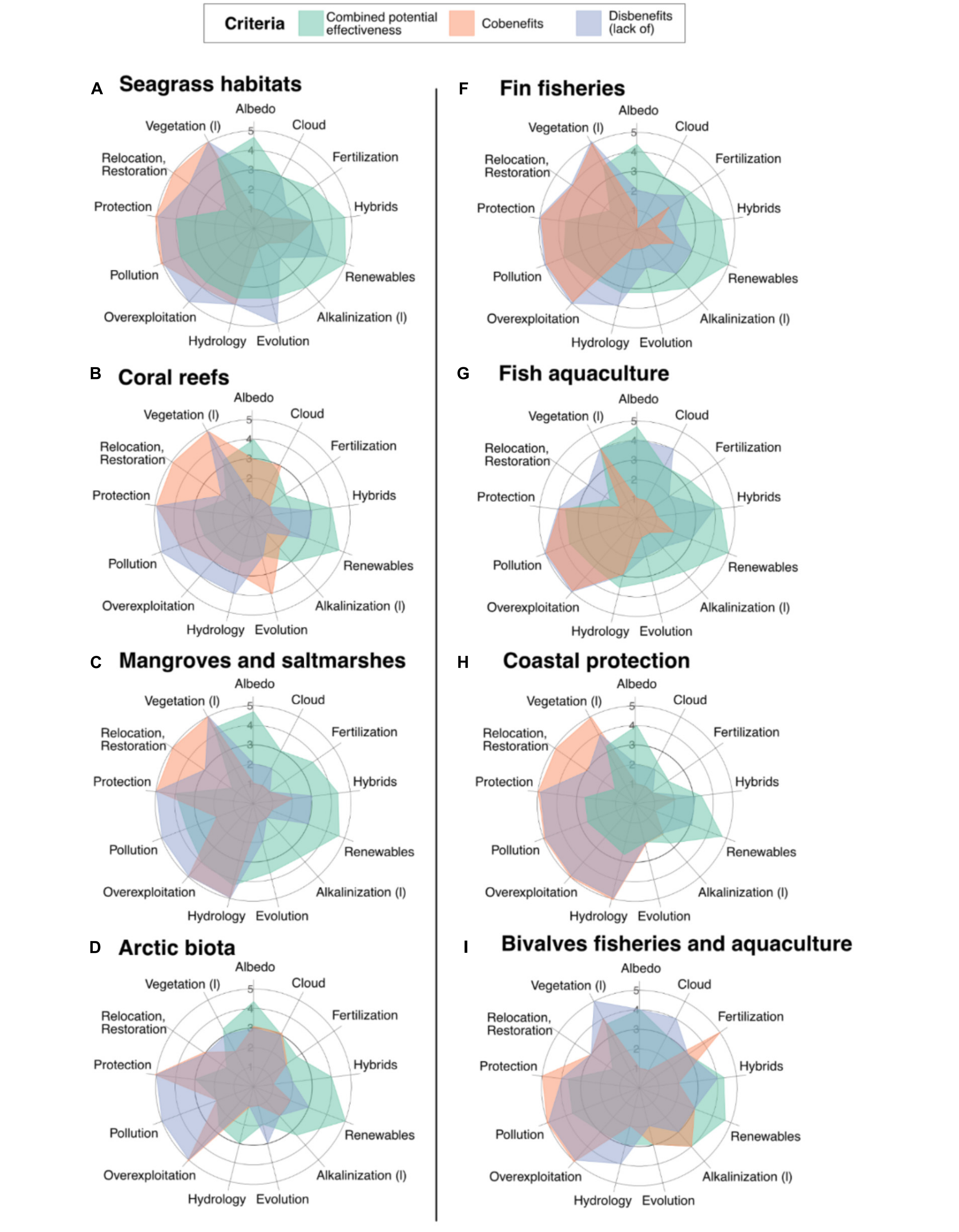

FIGURE 5 | Contribution of ocean-based solutions to reduce the impacts of key ocean drivers on key ecosystems **(A–D)** and ecosystem services **(E–H)**. The combined potential effectiveness represents the average potential effectiveness to reduce the impacts of ocean warming, ocean acidification, and sea-level rise (see section "SM3" of the **Supplementary Materials**). Scores 1 to 5: very low, low, moderate (thicker circle), high, and very high.

combined potential effectiveness; however, if reef restoration were considered separately from relocation, it would score higher (especially with regard to reducing local relative sea-level rise).

The potential to reduce the impacts of non-climatic drivers is a key attribute of local measures because it increases the resilience of ecosystems to climate change (O'Leary et al., 2017). For example, *protection* and *eliminating overexploitation* can support high reproductive outputs and juvenile recruitment following climate-related mass mortalities, allowing for population recovery from extreme events (Micheli et al., 2012; Roberts et al., 2017). Moreover, these measures produce co-benefits, such as spillover benefits of MPAs to adjacent areas supporting shellfish fisheries and aquaculture, and few, if any disbenefits, especially for coral reefs and vegetated marine habitats (Roberts et al., 2017). Some MPAs are more affected by coral bleaching than fished areas because they harbor more thermally sensitive corals (Graham et al., 2008) but there is a strong case that protected coral reefs recover better (Cinner et al., 2013).

Whilst local solutions can decrease the total (climate- and non-climate related) impacts and improve ecosystems' resilience, they cannot eliminate all of the climate-related component of impacts. For example, water quality and fishing pressure had minimal effect on the unprecedented bleaching of 2016 (Hughes et al., 2017b). Furthermore, and despite local protections, the changes associated with a high CO_2 emission scenario will result in further habitat and species losses throughout low-latitude and tropical MPAs, for example through the effects of warming on habitat-forming species such as corals, thereby reducing their beneficial roles (Bruno et al., 2018).

Despite the fact that most solutions implemented at local scales have a limited effectiveness to reduce the impacts of warming, acidification, and sea-level rise globally, they all have some beneficial effects, which could help in countering global climate impact if scaled beyond their current implementation. For example, seaweeds and seagrasses can reduce ocean acidification locally (e.g., Unsworth et al., 2012; Mcleod et al., 2013) and can potentially buffer adjacent coral populations by off-setting decreases in seawater pH (Camp et al., 2016).

POTENTIAL TO REDUCE IMPACTS ON ECOSYSTEM SERVICES

Sensitive ecosystem services are also expected to benefit from the implementation of measures that have the highest potential effectiveness in addressing climate drivers globally, such as *renewable energy* and *alkalinization* (**Figure 4A**). Our assessment, however, suggests that these measures may also lead to significant disbenefits (**Figures 5B–E**, sections "SM3.4 and SM3.4.3" of the **Supplementary Materials**). For instance, the addition of non-carbonate alkaline minerals may perturb biogeochemical processes though the release of mineral constituents such as cadmium, nickel, chromium, iron, and silicon (Hartmann et al., 2013). This may alter the pattern of primary and secondary production, and increase contaminant accumulation along the food chain (Russell et al., 2012; Alava et al., 2017), possibly impacting fisheries and aquaculture production, and the coastal protection value of coastal habitats. Furthermore, *alkalinization* is only moderately effective in reducing the impacts of sea-level rise, which is the primary driver affecting mangroves and saltmarshes. A similar conclusion applies to most of the global measures, notably *cloud brightening* and *albedo enhancement*, where large-scale deployment may risk high levels of disbenefits. In contrast, although our assessment suggests that large-scale *renewable energy* may lead to some local collateral damages on ecosystem services when these systems are deployed in coastal ecosystems, these impacts may be largely moderated through careful planning and consultation (Pelc and Fujita, 2002). In contrast, minimal damage is anticipated for deep-water floating systems currently being tested.

Measures that are most effective to reduce climate-related drivers locally (e.g., relative sea-level rise) often also have the dual benefit of minimizing the impacts from non-climatic drivers affecting coastal and marine ecosystems and environments (e.g., pollution, overexploitation, overfishing, and coastal development). As a result, the most effective local-scale interventions to maintain healthy conditions for fin fisheries, fish and bivalve aquaculture, and coastal protection are *eliminating overexploitation*, *restoring hydrology*, *reducing pollution*, *vegetation*, and *protection* (**Figure 5**). Modeling studies indeed suggest that the increase in stock abundance and productivity by effective management of fisheries and conservation of fish stocks (Costello et al., 2016) is likely to compensate losses from climate change (Cheung et al., 2017). It was shown that sustainable mangrove management interventions support surface elevation gains, thus limiting relative sea-level rise (Sasmito et al., 2016). More generally, *protection* and *vegetation* enable mangroves, saltmarshes, coral reefs, and seagrass to reduce the impacts of sea-level rise on coastal communities through wave attenuation and shoreline stabilization. Maintaining the health of ecosystems that provide coastal protection also has significant additional co-benefits to local human communities (e.g., carbon sequestration, water filtering, tourism, food security, recreation; Barbier et al., 2011; Weatherdon et al., 2016), in addition to supporting their resilience to climate impacts (Carilli et al., 2009). It is not surprising then that many countries are actively including marine ecosystems in their national climate plans as shown by the Nationally Determined Contributions submitted under the Paris Agreement (Gallo et al., 2017).

PATHWAYS TO IMPLEMENTATION

Clusters of Potential Solutions and Tradeoffs

A principal components analysis (see section "SM4" of the **Supplementary Materials**) was used to reduce the eight dimensions of our assessment dataset defined by the scoring criteria to two latent dimensions that explain most of the variance in the assessment data. Three clusters of schemes emerge (**Figure 6**). The first one includes *alkalinization* at the global scale, *hybrid methods, albedo enhancement,* and *cloud brightening*,

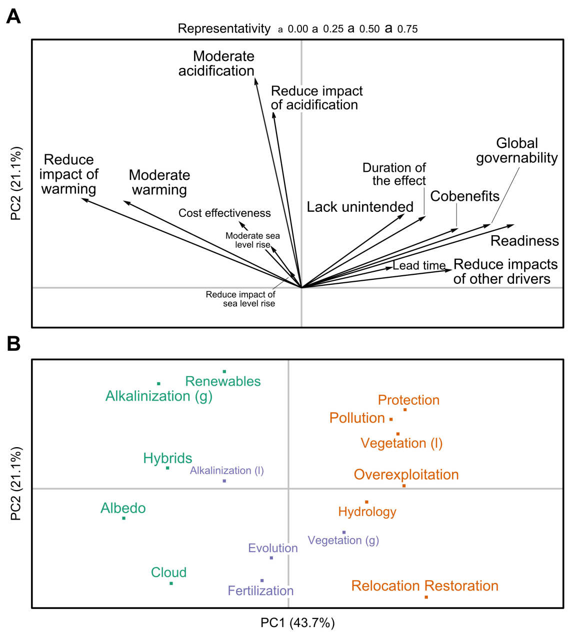

FIGURE 6 | Principal components analysis (PCA) of the attributes of ocean-based solutions. **(A)** Correlations among criteria, some being averaged across ecosystems and ecosystem services. When two arrows point in the same direction, the criteria are correlated: the scores of most solutions are similar for these two criteria (e.g., both warming-related criteria, co-benefits, and readiness). When they point in opposite directions, criteria are anti-correlated (e.g., moderate warming and global governability). When they are perpendicular, criteria are uncorrelated (e.g., acidification-related criteria and readiness). **(B)** Positions of solutions in the PCA. Solutions on the right have high scores in the criteria that point to the right and low scores in the attributes that point to the left; a similar reasoning can be made for any direction in this space. Solutions are clustered into three groups, through hierarchical clustering based on their position in the PCA space, and colored accordingly. The first two principal components explain 65% of the variance in the attributes of ocean-based solutions. Attributes or solutions that are not well represented in this space are shown in smaller font (representativity or "Repr." varies between 0 and 1). See section "SM5" of the **Supplementary Materials** for details on the PCA approach, and sections "SM3 and SM4" for additional information on the assessment.

which show high potential effectiveness to reduce warming and acidification, and their impacts. However, there has been relatively little research, testing and application on such solutions, and they generally score low for technological readiness, co-benefits, lack of disbenefits, and global governability. In contrast, the second cluster includes almost all local measures (*protection*, *reducing pollution*, *vegetation* at the local scale, *eliminating overexploitation*, *restoring hydrology* and *relocation and restoration*), and is characterized by low effectiveness to reduce warming and its impacts, and moderate effectiveness to reduce ocean acidification and relative sea-level rise and their impacts. These measures are, however, technologically ready, have significant co-benefits, few disbenefits and can also help to reduce the impacts of non-climatic drivers. *Renewable energy* stands apart as it exhibits both high potential effectiveness and technology readiness, thus ranging in between clusters 1 and 2. The third cluster includes *assisted evolution*, *alkalinization* at the local scale and *fertilization*, which have low to moderate scores across most criteria assessed.

Ocean Governance Challenges

Measures which are the most technically feasible also have the highest global governability (**Figure 4**). They comprise *protection*, *eliminating overexploitation*, *reducing pollution*, *vegetation*, *relocation and reef restoration*, and *renewable energy*. Except for the latter, all these measures are local. Their governability is high to very high except for *restoring hydrology* and *assisted evolution* (moderate or low). Global measures such as *albedo enhancement*, *fertilization*, *hybrid methods*, *cloud brightening*, and *alkalinization* have a lower overall technical feasibility, partly due to lack of testing and experience, together with moderate to low governability. Yet none of these schemes do much to reduce or moderate the impacts of the climate-related drivers considered in this study (ocean warming, ocean acidification and sea-level rise).

Such conclusions highlight the need for multiple-scale and multiple-stakeholder initiatives, hence calling for improved international governance mechanisms to ensure coherency in ocean-based climate action. These governance challenges are, however, constrained by controversies on the potential solutions, which scientific investigations and policy engagement can help overcome. Controversies are mostly in the "addressing the causes of climate change" and "solar radiation management" areas of action (**Figure 2**). They include: the moral hazard dilemma, i.e., that development and deployment of alternative solutions might decrease effort on emission reductions (Preston, 2013; McLaren, 2016); the risk of premature lock-in of suboptimal solutions and path dependencies (Burns et al., 2016; Reynolds et al., 2016); and concerns regarding controllability and transnational effects (Williamson and Bodle, 2016). Ethical issues are also important, relating to informed consent and potential adverse impacts on countries unable to deploy such measures (Svoboda, 2012; Suarez and van Aalst, 2017; Rahman et al., 2018); and vested interests, as production and deployment of innovative measures could be a highly profitable market (Preston, 2013). Controversies related to the "protection of biota and ecosystems" and "manipulation to enhance biological and ecological adaptation" areas of action mostly arise from conflicts relating to local, national and global-scale interests, and the balance between short-term and long-term benefits and disbenefits (Cooley et al., 2016; Cormier-Salem and Panfili, 2016). Such trade-offs between "winners" and "losers" highlight the influence of social norms and values that may differ greatly between different stakeholders (Hopkins et al., 2016; Lubchenco et al., 2016). Testing the veracity of such perceptions via further research and demonstration of novel measures at relevant scales will clarify governance issues.

The Way Forward

The global implementation or testing of *renewable energy*, *fertilization*, *vegetation*, *eliminating overexploitation*, and *protection* has accelerated sharply in the past two decades (**Figure 3**). In particular, several local measures (*vegetation*, *protection*, and *eliminating overexploitation*) may achieve their full potential in a few decades at their current rate of deployment.

Nevertheless, the scale of deployment for most solutions remains far below what would be necessary to effectively address climate change drivers and impacts (**Figure 4B**). Delivering the full potential of global measures such as *renewable energy*, *alkalinization,* and *hybrid methods* requires orders-of-magnitude increases in their research, testing, and deployment. Such action is considered urgent on the basis of the climatic threats to ocean sustainability (Gattuso et al., 2015), and since there are decadal lag times until full maximum effectiveness of all the global measures considered here (**Figure 4B** and section "SM3.1" of the **Supplementary Materials**). In the meantime, there will likely be significant increases in climate-related impacts on ocean ecosystems and services, which will reduce ecosystems' ability to provide local solutions (Albright et al., 2016; Cheung et al., 2016; Cinner et al., 2016), thereby decreasing leeway for action (Gattuso et al., 2015).

It is clear that the familiar and conventional marine management strategies cannot fully counter climate change and its impacts. Accelerating research and deployment of other potential solutions will, however, challenge the capacity of science, policy, and decision-making in evaluating and deploying solutions. Defining road maps to drastically enhance action faces major constraints relating to the large uncertainties in key non-climatic variables. Thus socioeconomic conditions may flip the balance between fossil fuel markets and renewables, potentially catalyzing a rapid acceleration of the deployment of marine renewables, but not necessarily with adequate consideration of local disbenefits. There is also a need to consider a broader range of measures than those assessed here, many of which are still in their infancy and unfamiliar to marine management (e.g., large-scale seaweed aquaculture, or abiotic methods of removing or stripping CO_2 from seawater). This calls for the development of policies and funding to foster and promote research into new or emerging ocean and climate management options.

OUTLOOK

Current pledges under the Paris Agreement are insufficient to hold the global average temperature increase to well below 2°C above pre-industrial levels, calling for a dramatic increase in global mitigation effort. However, even with a full and timely implementation of the Agreement, major impacts on sensitive marine ecosystems such as coral reefs and Arctic biota are expected, requiring additional, ambitious and rapid actions to address climate-related drivers locally, minimize their impacts, and increase resilience. To support efforts to address the ocean's potential contribution to these mitigation and adaptation goals, our assessment highlights five evidence-based key messages.

First, each measure has tradeoffs. For example, *alkalinization* scores high in global mitigation potential, but low in technological readiness or global governability. In contrast, measures implemented locally such as *protection* and *reducing pollution* have strong co-benefits and high governability, but have a much lower effectiveness to moderate changes in climate-related drivers. Decisions favoring any measure must therefore consider multiple criteria, including effectiveness, feasibility, co-benefits, disbenefits, governability, and cost effectiveness, rather than only the climate-related effectiveness or cost effectiveness.

Second, ocean-based measures with relatively high global effectiveness (such as *albedo enhancement)* have significant adverse side effects on key marine ecosystems and services. In contrast, local measures rank higher in terms of global governability, co-benefits and lack of disbenefits, and have a moderate ability to reduce climate-related impacts, only offering local opportunities for mitigation. The emerging picture is that actions in addition to local and more conventional marine management are needed to increase chances of avoiding or countering climate impacts. It is unlikely that a single measure will be able to meet a pathway consistent with the Paris Agreement. The introduction of multiple measures, including land-based ones, would require deployment of each of them at decreased scales relative to single-measure deployments, and would also reduce the risk of side effects (see also Minx et al., 2018).

Third, some measures that offer greater effectiveness in countering climate and its impacts (e.g., *alkalinization*, *cloud brightening*, *albedo enhancement*, and *assisted evolution*) currently exhibit too many uncertainties to be recommended for large-scale deployment until more research is conducted. However, measures with demonstrated potential effectiveness, co-benefits and with no or few disbenefits (*renewable energy* as well as other local solutions except *assisted evolution*) are no-regret measures that can be widely deployed immediately, as other potential solutions are explored. The high merits of *renewable energy* is consistent with the conventional policy approach that the best way to avoid climate impacts (on the marine environment, as well as elsewhere) is to eliminate the primary driver, excess atmospheric CO_2 concentration, by drastically reducing CO_2 emissions (Gattuso et al., 2015).

Fourth, climate change intervention at multiple scales requires that multiple and diverse actors are involved, hence calling for coordination across scales. Interestingly, besides being central to decisions on global measures, our assessment suggests that the international community can also play an indirect supporting role to the implementation of local solutions. The international community must therefore accelerate diplomatic and political efforts, especially within institutions such as the UNFCCC and the UN Convention on Biological Diversity, to improve existing arrangements or find new ones, and develop facilitative mechanisms for global to local action.

Fifth, since there are controversies and uncertainties on many of the measures we considered, a better scientific understanding of solution benefits, disbenefits, costs, and suitable governance arrangements is needed to inform policy and decision making. For example, 41% of the scores have low to very low levels of confidence (see section "SM3.4" of the **Supplementary Materials**). A major area of research thus relies in better determining potential effectiveness, cost-effectiveness, and desirability under various greenhouse gas emission scenarios. Furthermore, given the social challenges involved in all potential solutions, social science research is needed for understanding factors that hinder or promote effective and fair governance of ocean-based solutions (Magnan et al., 2016). In turn, this will allow a balanced consideration of new, unconventional ideas (e.g., regional cloud brightening to reduce pressures on coral reefs, advanced hybrid technologies, or innovative governance solutions for reconciling social conflicts associated to measures). This is a prerequisite for providing decision- and policy-makers with robust information, for example through the various products of the sixth assessment cycle of the IPCC. As new knowledge and insights become available, it is key that scientists effectively engage with the general public and decision makers, especially discussing the potential, feasibility, tradeoffs and social preferences of specific measures, and the consequences of failing to deploy solutions on time. This will notably help to increase mutual understanding and serve to reduce confusion and misinformation regarding the realized and future impacts of climate change on the ocean (Gelcich et al., 2014).

CONCLUSION

Both the marine policy and science communities need to recognize the uncertainties and limitations of currently available climate and ocean management options; support the immediate development of the most promising ones, e.g., *renewable energy* and local actions that can be scaled up; and acknowledge that new or emerging measures that are not part of current marine management practice might, through further research and testing, prove cost-effective as well as environmentally and socially acceptable.

AUTHOR CONTRIBUTIONS

J-PG, AKM, LB, WWLC, CMD, JH, EM, FM, AO, PW, RB, VIC, RDG, JJM, H-OP, and GHR designed and carried out the research. All co-authors conducted the analyses and wrote the paper.

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SUPPLEMENTARY MATERIAL

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frontiers in Marine Science

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Managing Recovery Resilience in Coral Reefs Against Climate-Induced Bleaching and Hurricanes: A 15 Year Case Study From Bonaire, Dutch Caribbean

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Coral reefs are among the world's most endangered ecosystems. Coral mortality can result from ocean warming or other climate-related events such as coral bleaching and intense hurricanes. While resilient coral reefs can recover from these impacts as has been documented in coral reefs throughout the tropical Indo-Pacific, no similar reef-wide recovery has ever been reported for the Caribbean. Climate change-related coral mortality is unavoidable, but local management actions can improve conditions for regrowth and for the establishment of juvenile corals thereby enhancing the recovery resilience of these ecosystems. Previous research has determined that coral reefs with sufficient herbivory limit macroalgae and improve conditions for coral recruitment and regrowth. Management that reduces algal abundance increases the recovery potential for both juvenile and adult corals on reefs. Every other year on the island of Bonaire, Dutch Caribbean, we quantified patterns of distribution and abundance of reef fish, coral, algae, and juvenile corals along replicate fixed transects at 10 m depth at multiple sites from 2003 to 2017. Beginning with our first exploratory study in 2002 until 2007 coral was abundant (45% cover) and macroalgae were rare (6% cover). Consecutive disturbances, beginning with Hurricane Omar in October 2008 and a coral bleaching event in October 2010, resulted in a 22% decline in coral cover and a sharp threefold increase in macroalgal cover to 18%. Juvenile coral densities declined to about half of their previous abundance. Herbivorous parrotfishes had been declining in abundance but stabilized around 2010, the year fish traps were phased out and fishing for parrotfish was banned. The average parrotfish biomass from 2010 to 2017 was more than twice that reported for coral reefs of the Eastern Caribbean. During this same period, macroalgae declined and both juvenile coral density and total adult coral cover returned to pre-hurricane and bleaching levels. To our knowledge, this is the first example of a resilient Caribbean coral reef ecosystem that fully recovered from severe climate-related mortality events.

Keywords: Bonaire (Dutch Antilles), coral reefs, coral bleaching, hurricanes, managed resilience, Caribbean, herbivory, resilience

INTRODUCTION

Large-scale and relatively recent coral mortality resulting in the collapse of coral reef ecosystems has been both conspicuous and well documented (Eakin et al., 2010; Hughes et al., 2018). Declines often result from climate-induced stresses such as acute El Niño warming or intense hurricanes. It is easy to conclude that the long-term prognosis for coral reefs is poor, but it is inaccurate to say there is nothing humans can do to halt or slow the decline of these beleaguered ecosystems. In fact, evidence in support of "managed resilience" (*sensu* Bruno et al., 2019) is central to this case study.

No management actions can "climate-proof" coral reefs. However, some management actions may improve the recovery of coral reef ecosystems following a climate-induced disturbance, and thus improve "recovery resilience". Resilience is an ecosystem property (Holling, 1973) that affects a system's ability to resist an extrinsic perturbation that fundamentally changes its structure or to recover from such perturbations (Gunderson, 2000).

We focus on the recovery of coral reefs because it has gone undocumented in some vast tropical regions (but see Ortiz et al., 2018). In a highly cited scientific study entitled: "*Disturbance and recovery of coral assemblages*" (Connell, 1997), all existing studies on patterns of recovery in coral reefs world-wide were reviewed. While over 50% of Indo-Pacific coral reefs including those in the Great Barrier Reef, Indonesia, Indian Ocean, Guam, and Hawaii were documented to have recovered, there were no examples of Caribbean coral reefs recovering from disturbance. Roff and Mumby (2012) updated this analysis with 38 studies since Connell's publication, yet only one small part of one Caribbean reef met the original criteria for recovery (Idjadi et al., 2006). To date, no Caribbean coral reef system comprised of many smaller reefs, has been shown to have recovered from a climate-induced perturbation.

Most published studies have documented the decline of coral reef ecosystems in the Caribbean (Gardner et al., 2003; Jackson et al., 2014) and in the tropical Pacific (Bruno and Selig, 2007). The global decline of coral reefs was the impetus for several high impact scientific papers with titles such as "*Confronting the coral reef crisis*" (Bellwood et al., 2004) and "*Rising to the challenge of sustaining coral reef resilience*" (Hughes et al., 2010), or specifically asking the shocking question: "*Are U.S. coral reefs on the slippery slope to slime?*" (Pandolfi et al., 2005). These alarming titles and the associated press coverage caught the attention of managers and policy makers, but, to date, there has been little progress operationalizing the management of coral reef ecosystems (i.e., managed resilience) to enhance recovery resilience. Nevertheless, some studies, such as, "*Capturing the cornerstones of coral reef resilience, linking theory to practice*" (Nyström et al., 2008), gave clear advice to managers:

> *"Moving toward operationalizing resilience theory is imperative to the successful management of coral reefs in an increasingly disturbed and human-dominated environment." (Nyström et al., 2008).*

Although coral reefs are complex ecosystems, relatively few "drivers" have been shown to be most important to their structure and function. "*Drivers*" are processes that control critically important aspects of coral reefs. Several processes can interact with one another that ultimately improve the recovery of coral reefs (Mumby and Steneck, 2008). For example, macroalgae (seaweed) are known to poison corals (Rasher and Hay, 2010) and reduce or halt the settlement and survival of juvenile corals (Arnold et al., 2010; Steneck et al., 2014). Since herbivorous fishes, such as parrotfishes and surgeonfishes, can reduce or eliminate macroalgae from coral reefs (Lewis, 1986; Williams and Polunin, 2001), they enhance coral recruitment and facilitate the growth of reef corals (Burkepile and Hay, 2008). Some branching corals create complex coral habitats into which juvenile reef fish recruit (Caselle and Warner, 1996). The resulting feedbacks maintain healthy coral reefs (Mumby and Steneck, 2008) and are thus the "cornerstones" for managing resilience as advocated by Nyström et al. (2008).

Coral reef "health" and resilience is complicated because all components interact. Therefore, it is difficult or impossible to define a specific ecosystem attribute (i.e., state variable) as being particularly healthy or unhealthy for any given coral reef. Instead, one must measure and track important components as they change through time (Edmunds et al., 2019). Indeed, there is a consensus on *trends* that constitute healthy trajectories in reef condition. Trends of increasing live coral cover or decreasing macroalgal abundance are both moving toward improved conditions. Herbivores reduce algal abundance but the amount of herbivory to be effective will vary from local to ocean basin scales. For example, coral reefs at identical water depths, light, and wave exposure have significantly greater rates of algal colonization and growth in the Caribbean compared to the tropical Indo-Pacific Ocean (Roff and Mumby, 2012). Accordingly, higher rates of herbivory are necessary to limit algal growth in the Caribbean. Managers perennially would ask how much herbivory is enough (Bozec et al., 2016)? Studying the trajectory of algal growth over time allows managers to determine the success or failure of strategies to manage herbivory or other factors that contribute to algal growth and/or success of reef corals.

Here, we report on a study initiated in 2002 on the island of Bonaire, Dutch Caribbean, in the southern Caribbean (**Figure 1**). For over 15 years, we regularly monitored the abundance of live coral, algae, reef fishes and juvenile corals at fixed locations at multiple sites, with the aim of documenting trends in key ecosystem drivers within the ecosystem. After 5 years of monitoring, Bonaire's coral reefs suffered two large-scale disturbances from a hurricane and a coral bleaching event. These disturbances were effectively a "*stress-test*" that allowed us to gauge the recovery resilience of the monitored coral reefs. All research was conducted in coordination and with the support of "Stichting Nationale Parken ('STINAPA') Bonaire," the environmental organization charged with managing Bonaire's coral reefs during the period of study.

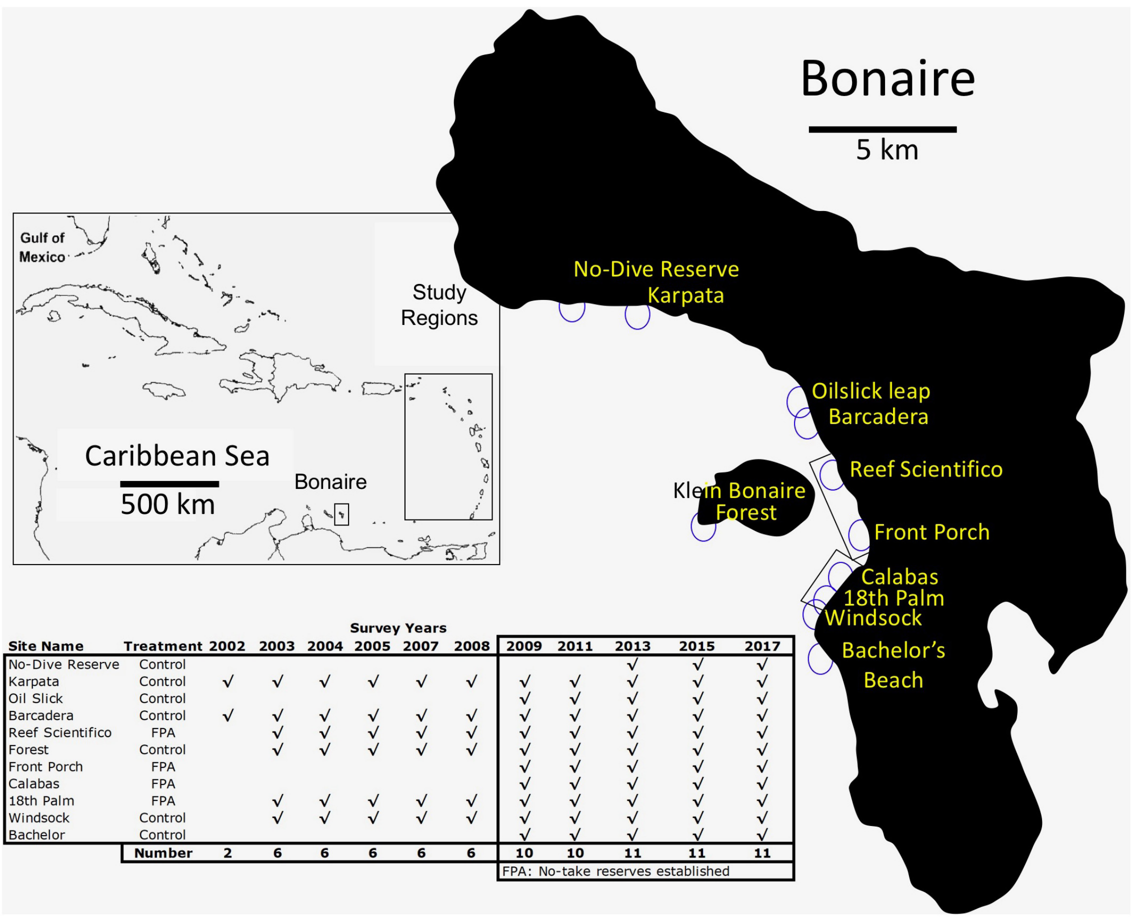
As is necessary with any single-location study we address the question, "compared to what"? Accordingly, we compared our long-term study of Bonaire's coral reefs with a recently published short-term but geographically broad study of the coral reefs of the Eastern Caribbean conducted in 2013–2014, that used identical methods at 12 islands over a 700 km area to contrast heavily fished sites with no-take reserves (Steneck et al., 2018). Like Bonaire, the reefs surveyed in the Eastern Caribbean were mostly in leeward environments and therefore comparable in terms of physical environment.

MATERIALS AND METHODS

Monitoring Sites and Protocols

In 2002 we began monitoring Bonaire's coral reefs at two sites selected by STINAPA as important and representative reef sites. Then in 2003, we began biannual monitoring expanding to six and then 11 STINAPA-selected sites (**Figure 1**). Monitoring was conducted during the first weeks of March during each monitoring year, so as to minimize seasonal variance.

At each site, we monitored multiple transects set at 10 m depth. At the six long-term monitored sites established in 2003, we placed 10 cm square tiles to create four permanent 10 m long transects per site for quantifying benthic organisms (methods detailed in Arnold et al., 2010). We added five new sites between 2009 and 2013, all set at 10 m depth. Since the reef slope is steep, the transect lines set during each monitoring period at



Site Name	Treatment	2002	2003	2004	2005	2007	2008	2009	2011	2013	2015	2017
		Survey Years										
No-Dive Reserve	Control									√	√	√
Karpata	Control	√	√	√	√	√	√	√	√	√	√	√
Oil Slick	Control							√	√	√	√	√
Barcadera	Control	√	√	√	√	√	√	√	√	√	√	√
Reef Scientifico	FPA		√	√	√	√	√	√	√	√	√	√
Forest	Control		√	√	√	√	√	√	√	√	√	√
Front Porch	FPA							√	√	√	√	√
Calabas	FPA							√	√	√	√	√
18th Palm	FPA		√	√	√	√	√	√	√	√	√	√
Windsock	Control		√	√	√	√	√	√	√	√	√	√
Bachelor	Control							√	√	√	√	√
	Number	2	6	6	6	6	6	10	10	11	11	11

FPA: No-take reserves established

FIGURE 1 | Bonaire, Dutch Caribbean. Inset rectangle shows the island location and the Eastern Caribbean sites from Steneck et al. (2018) used for comparison. Circles on the island map show the location of 10 m depth monitoring stations. Inset table shows duration and frequency of monitoring at those stations. Coastally located rectangles show the location of no-take reserves (called "Fish Protection Areas," or FPAs) that were established in 2008.

all sites were at, or very near, the specific benthic location of previous monitoring.

Coral and algal abundances were determined from four to eight replicate 10 m transects. Every organism or organism-group was identified under every cm of the transect. However, to only compare abundances on hard substrates, measurements taken from unconsolidated rubble or sand were removed. Each coral species and algal functional group (*sensu* Steneck and Dethier, 1994) were measured to the nearest cm. Our protocol was commensurable with the Atlantic and Gulf Reef Rapid Assessment (AGRRA) protocol (Lang, 2003).

Algal abundance was quantified by both measuring percent cover (via component lengths) and canopy heights (via a ruler, to the nearest mm) along the transects. This was done separately for filamentous algal turfs, macroalgae, and articulated calcified algae such as *Halimeda* spp. The percent cover of calcareous crustose coralline algae and less calcified peyssonnelid algae were quantified independently. We quantified the macroalga *Lobophora* spp. separately from other algae because it is known to be lethal to corals. We also calculated an algal biomass proxy using the volume of algae (called an "algal index" *sensu* Steneck et al., 2014) that is determined by multiplying algal percent cover by canopy height (mm).

Juvenile coral densities were quantified by placing a 25 × 25 cm quadrat at five locations where adult coral cover was ≤ 25% along each 10m transect (positioned 0, 2.5, 5, 7.5, and 10 m locations). Each coral species ≤ 40 mm maximum diameter was identified and measured to the nearest mm and counted for juvenile coral densities.

Fishes were quantified by swimming 7–10 replicate 30 m × 4 m belt transects at each site each year. This is a standard protocol used by reef scientists (methods in Steneck et al., 2018) and is commensurable with AGRRA. Fish surveys were conducted before benthic surveys to minimize human presence during fish surveys. For fish surveys, the observer attached a 30 m line to dead coral substrate and swam slowly to record all fish species and their sizes (to nearest cm) within the belt. During their return visit over the 30 m belt, the observer recorded the slower moving or less vagile fish taxa.

All of these methods and others along with results and data appendices were produced as biannual reports delivered to STINAPA Bonaire starting in 2003 (see **Supplementary Tables S1, S2** for benthic and fish data, respectively). The eight reports totaled 1001 pages and all are available from STINAPA Bonaire[1]. All data critical for this study are presented in this paper.

Benthic Community Structure Over Time

Data on the percent cover of individual coral species as well as the cover and canopy height of key algal functional groups were square root transformed to down weight the influence of the largest space occupiers. Temporal trends were then visualized using non-metric Multidimensional Scaling Ordination based on the Bray-Curtis dissimilarity coefficient (Clarke, 1993). First, an analysis of change was carried out to check that community structure varied over time. Here, site was included as a random effect and year as a fixed effect within a non-parametric analysis of variance, PERMANOVA (Anderson, 2001). The analysis was then repeated with specific contrasts of interest. These primarily included the key events in the time series; the impact of Hurricane Omar compared to prior years and the impact of the coral bleaching event compared to reefs post-Omar. Two additional comparisons were added for interest sake. The first compared the community structure in 2017 (7 years post-bleaching) to that of the first several years post-bleaching (2011–2015). This comparison was motivated by the rapid recovery observed by 2017, and because it was not specified *a priori*, we do not place importance on the statistical significance of that particular comparison; rather, we are interested to see which species drove that change. The final comparison asked whether the final community structure (2017) differed from that in the pre-disturbance years (i.e., prior to Hurricane Omar and the bleaching).

To understand which species drove the major changes in benthic community structure we used Similarity Percentage (SIMPER) analysis (Clarke, 1993). We confined our results to the five species or functional groups that mostly strongly accounted for the differences in community structure.

RESULTS

Trends in Coral, Algae, and Herbivores

Coral cover measured at 10 m depth at multiple sites (**Figure 1**) averaged around 45% until the first decline from Hurricane Omar (October 2008) was detected during the 2009 monitoring period (**Figure 2**). This was followed by the second coral mortality event resulting from the October 2010 bleaching event (Alemu and Clement, 2014). Following these two disturbances, live coral cover declined 22% from over 45% in 2007 to about 35% in 2011. Coral cover remained reduced until 2015 when it began to increase. By 2017, live coral cover had recovered to its pre-mortality abundance of over 47% (**Figure 2**). Note that even during the period of lowest coral cover in Bonaire, it exceeded the average found throughout the Eastern Caribbean (i.e., 17.8% ± 1.9 SE, see Steneck et al., 2018). The recovery in coral cover we documented for Bonaire in 2017 was sustained during the 2019 monitoring session. Coral cover recorded at the 11 sites was 46.3% (±2.8 SE) during the 2019 monitoring study (completed after the initial submission; see STINAPA website[1]).

Macroalgal abundance at all of Bonaire's monitored sites remained relatively low over the study period; nevertheless, there were important changes. Percent cover of macroalgae (the most common metric of algal abundance) varied between 1 and 6% cover at all sites from 2002 through 2005 (**Supplementary Figure S1**). Then, in 2007, average algal cover increased to 8% (±2.3 SE) and peaked in 2011 at 17.7% (±2.6 SE) cover. Algal biomass proxy ("algal index") increased sharply in 2011 following the coral bleaching event but then declined during each subsequent monitoring period (**Figure 3**). By 2017, percent cover of macroalgae retreated back to the pre-disturbance level of 6% (±1 SE) although canopy heights remained relatively high

[1]https://stinapabonaire.org/nature/research-reports/coral-reefs-adjacent-waters/

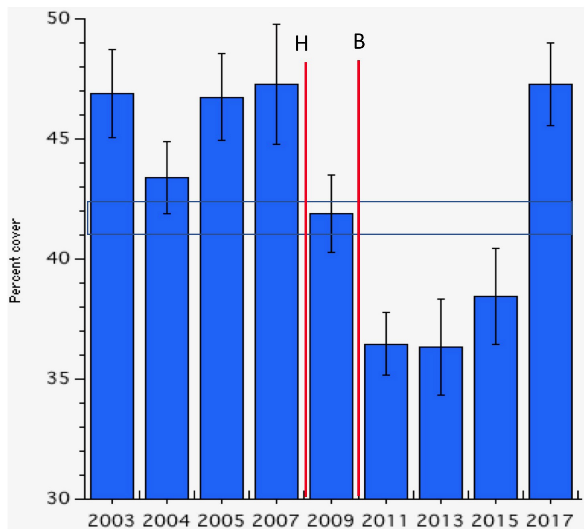


FIGURE 2 | Mean percent cover of live coral at monitored sites (variance here and throughout is expressed as standard error). The horizontal lines illustrate one standard error of the mean over the period. Vertical lines labeled "H" and "B" denote Hurricane Omar (October 2008) and the severe coral bleaching event of October 2010, respectively. For comparison, the average coral cover in the Eastern Caribbean was (17.6% ± 1.9 SE; Steneck et al., 2018 and see below).

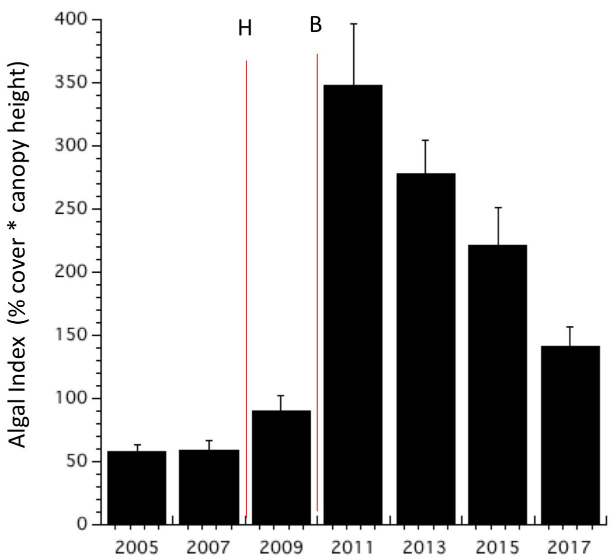


FIGURE 3 | Algal biomass proxy, called the "algal index" (percent cover times canopy heights for all algae) at monitored sites from 2005 to 2017 (mean ± SE). For comparison, the mean algal index across 12 islands of the Eastern Caribbean (i.e., 882 ± 58 SE; Steneck et al., 2018) was more than twice the maximum value recorded for Bonaire in 2011. Vertical lines denote the hurricane (H) and bleaching (B) events.

(**Supplementary Figure S2**). Nevertheless, the average percent cover of macroalgae at all study areas of the Eastern Caribbean was 25%, four times higher than that recorded for Bonaire (Steneck et al., 2018).

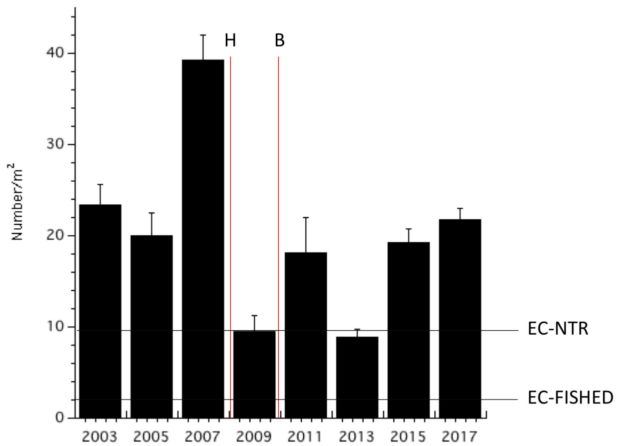


FIGURE 4 | Mean juvenile coral densities from 2003 to 2017 (results are from 316 transects and 1580 quadrats). Vertical lines showing hurricane and bleaching impacts are labeled as in **Figure 2**. Note the increase in 2011 was due to recruitment of *Agaricia* spp. that then grew out of the juvenile size class and contributed to the overall increase in coral cover.

Juvenile coral densities were relatively high throughout the study period (mean: 21.7 per m^2 ± 1.12 SE, from 440 quadrats along 88 transects since 2003; **Figure 4**). This is high when compared to the juvenile coral densities found in the Eastern Caribbean, which ranged from 1 to 10 among fished and no-take reserve sites (**Figure 4**; Steneck et al., 2018). Over the 2003–2017 study period, juvenile coral densities below 10 juvenile corals/m^2 were only observed in 2009 and 2013 (**Figure 4**).

An analysis of spatiotemporal trends in juvenile coral density unsurprisingly found that 'year' had the greatest impact, explaining 26% of the variance (p = 0.001). The planned temporal contrasts found significant effects of disturbance (i.e., 2005/2007 pre-disturbance vs. 2009/2011) and differences between the pre-disturbance density (2005/2007) and that of the final year. These impacts accounted for a further 9 and 3% of the variance respectively (p = 0.001). Macroalgal cover was negatively associated with juvenile coral density and explained 3% of the variance (p = 0.005) on its own. However, significant interactions occurred between time and macroalgal cover. The overall interaction explained 5% of variance (p = 0.011) and the specific interactions, macroalgal cover vs. disturbance effect and macroalgal cover vs. pre/post disturbance, accounted for 4% (p = 0.006) and 1% (p = 0.02) respectively. There was no effect of macroalgal cover pre-disturbance but a significant negative effect post-disturbance. Site was entered as a random effect but was not significant (p = 0.57).

Bonaire's reef corals at 10 m depth were strongly dominated by species of the major framework-building coral *Orbicella* species (**Figure 5**). The two dominant coral species of that genus combine to about 20% cover. The decline in coral cover following the two disturbance events was sharpest in *Agaricia agaricites*, *Madracis,* and *Colpophyllia* (**Figure 6**). These declines were added to the gradual

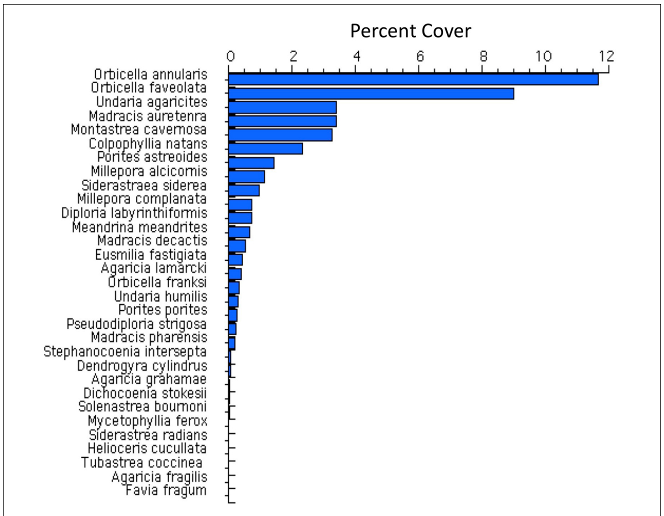


FIGURE 5 | Average percent cover of coral species from all 307 transects quantified from 2004 (when we began documenting individual species) to 2017.

declines in the two dominant species *O. annularis* and *O. faviolata*. *Montastraea cavernosa* increased in abundance over the early post-recovery study period. All of the six dominant species increased in the most recent monitoring periods (**Figure 6**).

Although all major species of reef fishes were censused regularly (see STINAPA website[2]), here we report only on scarine (parrotfish) populations. Parrotfish have been shown to be the most abundant and most important herbivores on Caribbean coral reefs and can facilitate coral recruitment by reducing algal biomass (Mumby et al., 2006; Arnold et al., 2010; Mumby and Harborne, 2010; Steneck et al., 2014).

Parrotfish biomass and population densities declined from 2003 to 2009 (**Figures 7**, **8**). Both biomass and abundance then stabilized until 2017 when parrotfish densities increased sharply. Parrotfish biomass recorded in 2017 was twice that recorded throughout the Eastern Caribbean, including the no-take reserves (i.e., "EC-NTR" in **Figure 7**). Parrotfish biomass did not change following the hurricane and bleaching events (**Figure 7**). In contrast, parrotfish population density declined in 2011 and remained low by Bonaire's standards until 2017 (**Figure 8**). Monitoring should be continued to be sure parrotfish abundance remains sufficiently high to control algal abundance.

Benthic Community Structure Over Time

The benthic community structure varied significantly over time and space ($p = 0.001$; **Figure 9A**). Specifically, each of the major

[2]https://stinapabonaire.org/nature/research-reports/coral-reefs-adjacent-waters/

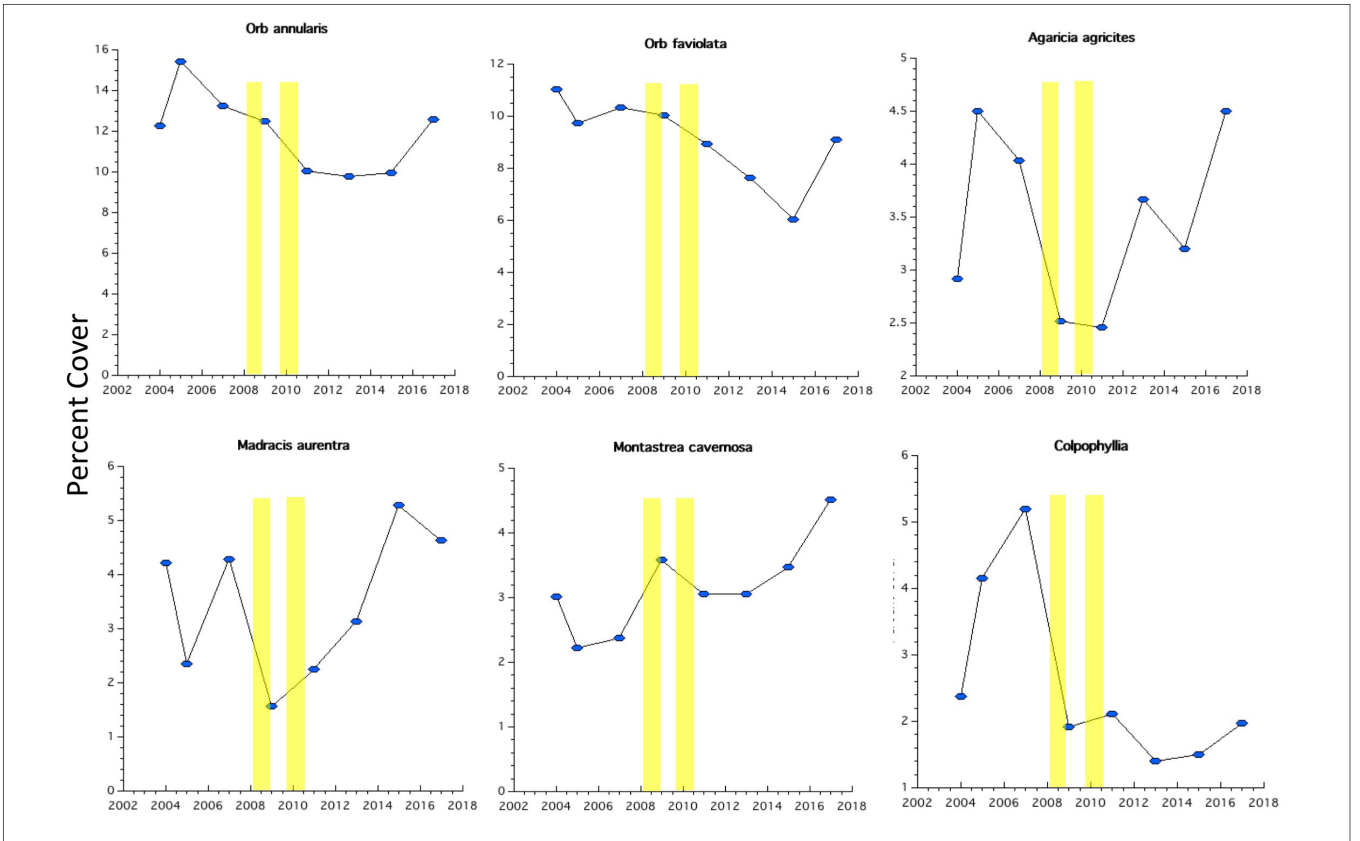


FIGURE 6 | Abundance trajectories of the six most abundant coral species since 2004. Vertical yellow bars denote the timing of Hurricane Omar and the 2010 coral bleaching event.

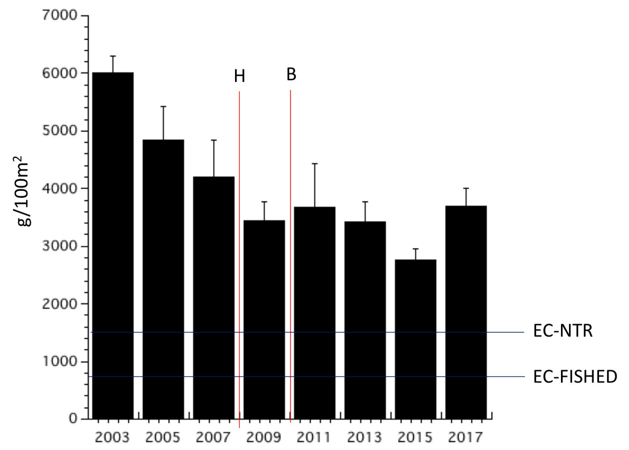


FIGURE 7 | Mean parrotfish biomass from repeated fish transects (+SE). The horizontal lines represent the mean values for the Eastern Caribbean no-take reserves ("EC-NTR") and fished areas ("EC-Fished") studied in Steneck et al. (2018).

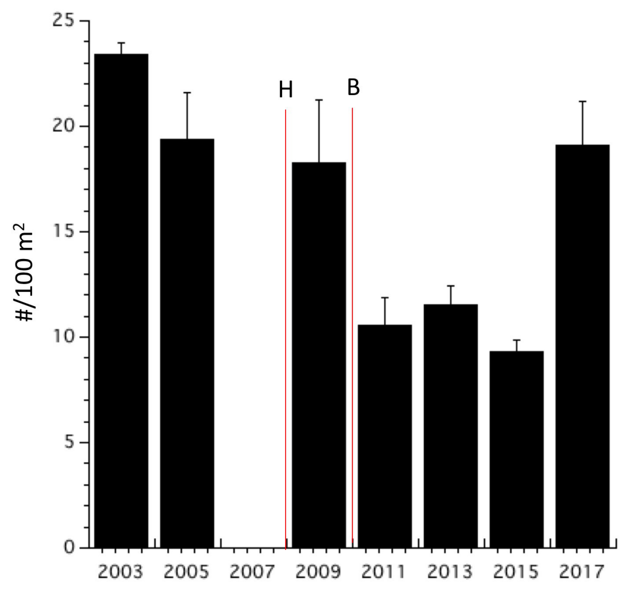


FIGURE 8 | Parrotfish population density (mean ± SE). Notations as in **Figure 2**.

periods of reef state differed from one another (p = 0.001). The hurricane led to important declines in *Orbicella faveolata* and *O. annularis*, though *Montastraea cavernosa* was able to increase during this period (**Figure 9B**). Not surprisingly, macroalgal cover and canopy heights also increased following the hurricane. Overall, the impact of the hurricane explained 9% of the total spatio-temporal variance of benthic community structure (**Figure 9C**). The advent of bleaching post-hurricane led to further changes in the community, characterized by further losses in the two *Orbicella* species listed as well as a reduction in *M. cavernosa* and continued increases in macroalgal cover and canopy height. Yet the impact of the bleaching event only explained 4% of the total variance. By 2017 coral cover had recovered to pre-disturbance levels (**Figure 2**), but the community structure had not yet returned to earlier compositions (p = 0.001, **Figure 9A**). Indeed, the difference in community structure between pre-disturbance and 2017 accounted for 12% of total spatio-temporal variance (**Figure 9C**). Looking specifically at the final periods of reef recovery that occurred between the post-bleaching years 2011–2015 and 2017, we found that the change in benthos was characterized by increases in the major framework builders *Orbicella* and *Montastraea* with a concomitant decline of macroalgae (**Figure 9C**). This final period of recovery accounted for 6% of total variance. Note that 46% of all the variance in benthic community structure could not be accounted for in terms of the major disturbance and recovery events.

DISCUSSION

Bonaire's Resilient Coral Reefs

Bonaire's coral reefs are characterized by having much higher parrotfish biomass, lower abundance of macroalgae and higher coral cover than most Caribbean coral reefs (**Figures 2–10**). In 2017, Bonaire's coral and algal cover were at or better than the Caribbean average from the 1970's (**Figure 10**; Jackson et al., 2014). Also, Bonaire's coral species composition is a throwback to the structure of coral reefs past (with the exception that acroporids remain lacking). For example, while most coral reefs throughout the Caribbean in recent decades have been dominated by "weedy" coral species such as *Porites astreoides, P. porities, Agaricia* spp., and *Siderastrea* spp. (Pandolfi and Jackson, 2006; Steneck et al., 2018; Bruno et al., 2019), the dominant corals on Bonaire's reefs were the massive reef building corals of the past such as *Orbicella annularis, O. faveolata,* and *Montastraea cavernosa* (**Figure 5**). The relatively delayed recovery among the massive reef building *Orbicella* corals (**Figure 6**) compared to the rapid recovery in the "weedy" *Agaricia* sp. (**Figure 6**) contributed to the low proportion of variance explained post-disturbance (**Figure 9**). However, all of the massive reef building corals were increasing in coral cover by the most recent 2017 study (**Figure 6**). In sum, Bonaire's coral reefs appear to have uniquely resisted the changes that have swept through the Caribbean and have shown the capacity to recover (**Figures 2–8**). Therefore, Bonaire's coral reefs are, by definition, resilient - but why?

What is unique about Bonaire's reefs and their management? Clearly, factors such as low hurricane frequency in the Southern Caribbean and limited terrestrial runoff due to the island's low rainfall and low relief have likely contributed to Bonaire's reef condition. Moreover, the fact that the reefs experience low levels of wave exposure – they occur on the leeward side of the island – also suggests that macroalgal growth rates should be lower than those found in windward systems such as Belize, Florida, and Jamaica (Renken et al., 2010). However, those factors alone have not been enough to confer recovery resilience elsewhere in the Caribbean.

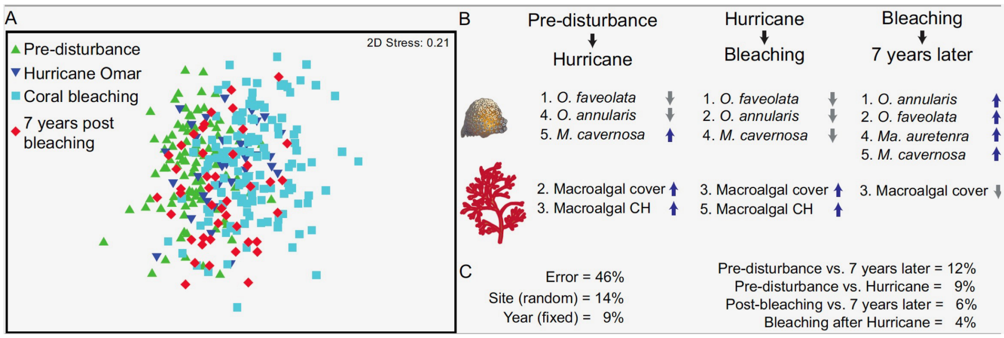


FIGURE 9 | Temporal changes in benthic community structure for the major periods of disturbance and recovery. Visualization is provided using a non-metric MDS **(A)**. The top five most important taxa characterizing benthic changes **(B)** are ordered in declining sequence (i.e., 1 being most important), and a breakdown of variance components is provided **(C)** for the total spatio-temporal variability in community structure, including the contrasts.

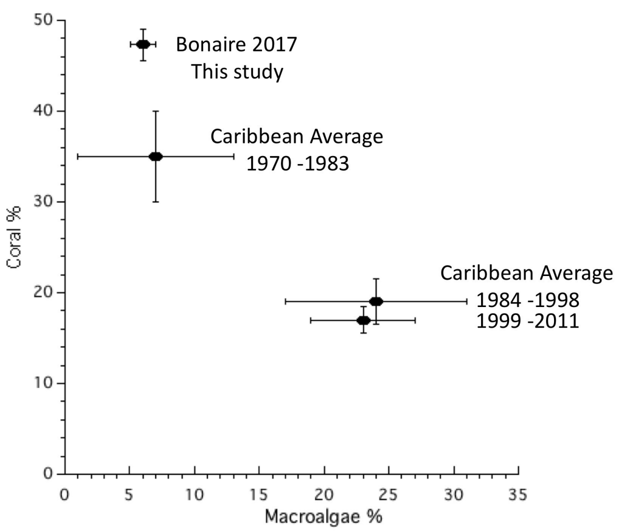


FIGURE 10 | Comparison of mean coral and macroalgal abundance for Bonaire and the rest of the Caribbean between 1970 and 2011 (from Jackson et al., 2014).

Effective and Sustainable Managed Resilience

The recent history and management of Bonaire's coral reefs is unique and worthy of specific consideration because social, ecological and economic synergies may have evolved that resulted in uniquely effective but also sustainable coral reef management. Starting in the 1960s, scuba diving on Bonaire's coral reefs became the island's biggest tourist draw and economic engine. By 1994, nearly half of the 57,000 tourists who visited the island were scuba divers (De Meyer, 1998). In 1971 the island banned spearfishing (**Table 1**). We are not aware of any other island in the Caribbean (or elsewhere) having adopted that regulation at that time. The spearfishing ban preceded the 1979 establishment of the Bonaire National Marine Park (BNMP) managed by STINAPA Bonaire. STINAPA Bonaire is a non-government organization founded with the vision that: "nature is recognized and treasured as the main resource of Bonaire's existence and sustainable development."[3]

Over recent decades, Bonaire's coral reefs became a favored destination within the scuba diving community because the reefs were known to be in relatively good condition and almost all dive sites were accessible from shore. Diving on coral reefs with abundant coral and fishes continues to be the island's primary tourist draw (Uyarra et al., 2005). This also created a unique funding opportunity because STINAPA Bonaire is a Non-Government Organization (NGO) that began charging a fee for scuba diving in 1992. The user fee funds Bonaire's National Marine Park management (i.e., >90% of STINAPA Bonaire's budget is covered by the diving user fee; Solofa, 2017). Analysis of the diving user fee determined it was a good business model for sustainable financing for Bonaire's marine conservation (Thur, 2010).

Management efforts often fail unless they conform with socioeconomic needs. Several factors contributed to Bonaire's management success. The early and successful development of the dive and hotel industries became the island's economic engine. In recent years, most economically important fishing targeted pelagic fish such as mahi mahi, tuna, and wahoo rather than coral reef dwelling fishes (Nenadovic, 2007). The relatively vibrant tourism-driven economy meant that relatively few people in Bonaire depended on fishing coral reef fishes for food.

Managing for Resilience, Monitoring for Trends

Overall, fishing pressure on Bonaire's coral reefs is relatively low and management actions have actively reduced emerging threats. Traditional shore-based hook and line fishing and regulations that restricted destructive practices such as anchoring and spearfishing had been banned years ago (Nenadovic, 2007). Fish traps, that naturally capture and kill parrotfish (Hawkins et al., 2007), were rarely used in Bonaire, but once they began being used (Nenadovic, 2007) legislation was passed in 2010 to phase them out along with a complete ban on the harvest of parrotfishes (**Table 1**). The cumulative results of these unique traditions and

[3]https://stinapabonaire.org/stinapa/

TABLE 1 | Summary timeline of Bonaire's coral reef management actions and environmental events.

Year	Management history	Environmental and reef events	Notes
1971	Banned spearfishing		
1975	Banned harvest of live coral		
1979	Bonaire National Marine Park established		Entire coast to 60 m depth (2700 hectares)
1980s	Acropora mass mortality. "STINAPA Bonaire" is established as a separate NGO foundation.		Part of Caribbean-wide decline
1992	Nature Tag (fee) funds STINAPA Bonaire NGO for reef management		Accounts for 93% of the management budget
1999	First AGRRA survey	Hurricane Lenny (first hurricane since 1877) (De Meyer, 1998)	
2003	Regular monitoring begins		
2008	No fishing areas established ("Fish Protection Area" or FPA).	Hurricane Omar October 2008	
2009	Declining parrotfish, juvenile corals and increasing algae	1st Lionfish (October 2009)	
2010	Fishing for parrotfish banned, fish traps phased out	Massive coral bleaching event	Bonaire referendum votes to join Netherlands

long-term management practices resulted in Bonaire's unusually abundant parrotfish, low macroalgal cover, and abundant coral (**Figures 2–4**, **7–9**). However, rather than "grazing intensity [being] sufficiently high that macroalgal blooms are prevented" (Mumby and Steneck, 2008) we found levels of herbivory were sufficient to reverse a small macroalgal bloom over time.

Bonaire's long duration of reduced fishing pressure on parrotfish likely resulted in the island's unusually abundant parrotfish populations. This created grazing pressure that could have prevented an earlier phase shift to macroalgae when the herbivorous sea urchin *Diadema antillarum* suffered a mass mortality throughout the Caribbean (1983-1984; Lessios et al., 1984). During that time, many Caribbean reefs were under intense fishing pressure (for numerous fish species, including parrotfish). Without fish predators or herbivore competitors, *Diadema* populations expanded to become the primary herbivore on most heavily fished reefs (Hay, 1984). Following the mass mortality of the sea urchin, the heavily fished coral reefs rapidly became macroalgal-dominated (Hughes, 1994; Steneck, 1994). However, there is no record of Bonaire having abundant *Diadema* or becoming algal dominated after the sea urchin mass mortality event (Van Duyl, 1985). Even if parrotfish had been relatively rare prior to the 1971 ban on spearfishing, more than a decade had passed prior to the sea urchin decline. Since a study in Bermuda determined parrotfish abundances can recover in less than a decade (O'Farrell et al., 2015), it is likely that Bonaire's parrotfish were sufficiently abundant to have prevented the macroalgal phase shift that swept through most of the rest of the Caribbean in the 1980s.

Knowing the history of an ecosystem can help interpret its current state. Measurements of state variables such as the percent cover of live coral, macroalgae, and the abundance of reef fishes are often taken as indicators of the condition of coral reefs. In fact, based on those variables, Bonaire was identified as one of three coral reefs in the tropical western Atlantic determined to be in "better" condition following a 1-year, Caribbean-wide assessment of coral reefs (Kramer, 2003). Nevertheless state variables, when taken as a "snap shot," cannot depict the trajectory or resilience potential of a coral reef ecosystem. What is needed is to determine the processes that drive those state variables.

The "two faces of resilience" (Holling, 1996) integrate both the capacity of an ecosystem to resist change or its capacity to recover to its previous state following a disturbance (Gunderson, 2000; Mumby and Steneck, 2011). Management actions generally cannot prevent coral mortality from coral bleaching, hurricanes or disease which means that studies that fail to detect an effect of reserves on coral resistance (Bruno et al., 2019) should not be a surprise. Rather, managed resilience primarily focuses on processes contributing to the recovery of coral reefs. Assuming a reef is not 'starved' of coral larvae (e.g., Hughes et al., 2019), the fundamental drivers facilitating coral recruitment and regrowth include (but are not limited to) ecological processes that limit algal production.

Numerous studies determined abundant algae impedes or halts coral recruitment (Birkeland, 1977; Arnold et al., 2010; Hughes et al., 2010; Steneck et al., 2014), therefore factors limiting algal biomass are fundamental to the recovery resilience of coral reefs. Herbivory resulting in relatively algal-free Caribbean coral reefs in the 1970s (**Figure 10**) was maintained by both herbivorous sea urchins and parrotfishes (Hay, 1984). However, following *Diadema's* mass mortality in the 1980s, parrotfish became the primary herbivore limiting algae throughout the Caribbean (Mumby, 2006). While today parrotfish is perhaps the most important herbivore group in the Caribbean, that is certainly not the case globally. Thus, there is no applicable circumtropical "parrotfish paradigm" (as asserted by Bruno et al., 2019) because a diversity of other herbivores (including acanthurids; Mumby et al., 2016) and other functional groups of fishes (Bellwood et al., 2006) can limit algae and thus indirectly drive coral reef recovery in tropical Indopacific coral reefs. Further, because the abundance of algae reflects both the rates of production and consumption of algae (Steneck and Dethier, 1994), lower rates of algal-clearing herbivory are necessary on reefs having a lower productivity potential such as tropical Indopacific coral reefs (Roff and Mumby, 2012).

Prior to our study, no systematic monitoring had been conducted on Bonaire's coral reefs. The mass mortality of the relatively shallow (<10 m) acroporid corals in the 1980s and Hurricane Lenny in 1999 damaged coral reefs both shallower and at locations other than our monitored reefs (**Figure 1**). The greatest perturbations to Bonaire's monitored coral reefs occurred between 2008 and 2010, with coral mortality from Hurricane Omar and the coral bleaching event, respectively. These disturbances resulted in a 22% decline in coral cover which is the greatest documented acute decline in Bonaire's history (**Figure 2**). When coral dies, it is quickly colonized by benthic algae. Because algae respond most rapidly to the increased surface area created by the recently dead coral, herbivore per area bite rates declined. Over time, as coral cover increased (**Figure 2**), herbivory may once again become concentrated to pre-disturbance levels (**Figure 3**). By 2017 we observed a modest increase in parrotfish density and adult and juvenile coral abundances and declining macroalgal abundance (**Figures 2–7**). Collectively, these trends represent the first example of complete resilience of a coral reef ecosystem in the Caribbean.

Social-Ecological Feedbacks and an Uncertain Future

The stability of alternative states depends on the ecological and social feedbacks that either facilitate or inhibit key drivers of ecosystem structure and function. Experiments that reduced grazing pressure from large parrotfishes resulted in increases in algal abundance and drove coral recruitment to zero (Steneck et al., 2014). Our study suggests that managing to maintain sufficiently high levels of herbivory to control algal abundance can create conditions that facilitate survival and growth in both juvenile and adult corals. We also suggest that socio-economic feedbacks that minimize the need to fish on coral reefs for food contributed to the recovery resilience of this coral reef ecosystem. It is also important that tourists who flock to Bonaire to view healthy reefs pay for the management necessary to keep them in that condition.

While we found complete recovery occurred 8–10 years following climate-driven disturbances, we hasten to point out that both hurricane and coral bleaching events have been increasing in frequency in recent decades. The Southern Caribbean has naturally low hurricane frequency. Prior to Hurricane Lenny in 1999, the last major hurricane to hit Bonaire was in 1877 (De Meyer, 1998), however Hurricane Omar struck 9 years after Lenny. Climate change is certainly increasing the frequency, strength, and size of hurricanes (Knutson et al., 2010), and thus, the frequency of collisions with Bonaire will likely increase as well. Coral bleaching throughout the tropics has been increasing in recent decades (Hughes et al., 2018). Moreover, even in relatively well-managed systems like the Great Barrier Reef, coral recovery rates have been declining in recent decades because of a combination of water quality (MacNeil et al., 2019) and the legacies of major disturbances which have a disproportionate impact on subsequent recovery rates (Ortiz et al., 2018). Therefore, concerns continue that even the best managed coral reefs may be unable to recover in a world growing increasingly hostile to these ecosystems (Hughes et al., 2018). Nevertheless, there is much to learn from the management successes of Bonaire's coral reefs. They demonstrate that local management today can contribute to the recovery resilience of this endangered ecosystem.

AUTHOR CONTRIBUTIONS

RS designed the monitoring protocol and drafted the manuscript. PM contributed to concepts and methods to study reef fishes. PM and RS analyzed the data. All coauthors conducted field work, monitoring data, analyses, and final report writing.

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SUPPLEMENTARY MATERIAL

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Lessons From the Pacific Islands – Adapting to Climate Change by Supporting Social and Ecological Resilience

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By necessity, Pacific Islands have become hubs of innovation, where climate strategies are piloted and refined to inform adaptation efforts globally. Pacific Island ecosystems are being degraded by pollution, overfishing, and unsustainable development. They also increasingly face severe climate impacts including sea-level rise, changing temperature and rainfall patterns. These impacts result inchanges in food and water security, loss of identity, climate-induced migration and threats to sovereignty. In response, communities in the region are leading climate adaptation strategies, often combining traditional practices and cutting-edge science, to build the resilience of their communities and ecosystems in the face of increasing climate risk. For example, communities are implementing resilient networks of marine protected areas using the best available science and strengthening tribal governance to manage these networks, experimenting with salt and drought tolerant crops, revegetating coastlines with native salt-tolerant plants, revitalizing traditional wells, and implementing climate-smart development plans. Often these efforts contribute to local development priorities and create co-benefits for multiple sustainable development goals (SDGs). These community efforts are being scaled up through provincial and national policies that reinforce the critical role that ecosystems play in climate adaptation and provide a model for the rest of the world. While adaptation efforts are critical to help communities cope with climate impacts, in some cases, they will be insufficient to address the magnitude of climate impacts and local development needs. Thus, there are inherent trade-offs and limitations to climate adaptation with migration being the last resort for some island communities.

Keywords: small island developing states (SIDS), climate change, Pacific Islands, vulnerability, adaptation, ecosystem-based adaptation

INTRODUCTION

The Pacific Islands are facing devastating impacts of climate change including increasing droughts and water scarcity, coastal flooding and erosion, changes in rainfall that affect ecosystems and food production, and adverse impacts to human health (IPCC, 2014, 2018).

Overpopulation, pollution and overuse of natural resources (e.g., overfishing and intensive land and water use), and unsustainable development and mining are also degrading island ecosystems

(Burke et al., 2011; Hills et al., 2013; Balzan et al., 2018). While the Pacific Islands are often described as highly vulnerable to climate change and lacking adaptation options (Pelling and Uitto, 2001), such descriptions disregard the ways in which Pacific Islanders are leading climate action and combining their own systems of knowledge with western science to implement locally relevant climate solutions (Barnett and Campbell, 2010; Mcleod et al., 2018). The lack of appreciation for Pacific climate leadership is exacerbated by biases in climate research that prioritize western science and technological solutions over other systems of knowledge (Jasanoff, 2007; Alston, 2014). It is critically important for global climate policy and national governments to recognize and support community efforts to build resilient communities and ecosystems through ecosystem-based adaptation strategies that are rooted in traditional knowledge and reinforced and supported by climate science, traditional leadership structures, and sustainable climate solutions.

Pacific Island leaders, along with leaders from other Small Island Developing States (SIDS), have been instrumental in shaping climate policies and the Paris Climate Agreement (UNFCCC, 2015). They called for a loss and damages clause that allows islands to assess and quantify impacts of cyclones and weather-related events and were vocal advocates to limit warming of global mean temperature to 1.5°C. The recognition that warming of 1.5°C or higher increases the risk associated with irreversible damages such as the loss of entire ecosystems has just been articulated in the latest IPCC report (IPCC, 2018). Despite their minimal contribution to global greenhouse gas emissions (Hoad, 2015), many SIDS included ambitious mitigation targets in their national climate plans (i.e., Nationally Determined Contributions, NDC) to raise collective ambition to reduce GHG emissions globally (Ourbak and Magnan, 2018).

Pacific Islanders are also leading climate action at the local level, implementing strategies to help communities and ecosystems to be more resilient to climate change. The region provides important opportunities for testing and refining adaptation responses at scale. The Pacific Islands are home to species found nowhere else on earth and are incredibly diverse, in terms of their ecosystems, geography, and demographics. Pacific Islanders have lived with natural environmental impacts for thousands of years and have adapted practices to accommodate periods of environmental fluctuations. Although the pace of environmental and climatic changes has increased, many communities are implementing climate-smart agriculture and are revitalizing traditional practices that utilize drought-tolerant species and the benefits of nature, such as using seaweed as compost to make soil more fertile, using palm fronds to shade plants during droughts, and planting vegetation to reduce flooding and erosion along coastlines. They are also combining these traditional practices with new scientific advancements such as the development of salt-tolerant and heat-tolerant crops and community-led GIS mapping of breadfruit trees vulnerable to climate impacts in the Marshall Islands. Communities are revitalizing traditional wells, establishing new protected areas and improving the management of existing protected areas, and developing climate-smart development plans that incorporate ecosystem-based adaptation.

However, ecosystem-based adaptation (EBA) efforts initiated by Pacific Island communities have largely been ignored in the peer-reviewed literature. Ecosystem-based adaptation is defined as combining biodiversity and ecosystem services into an adaptation and development strategy that increases the resilience of ecosystems and communities to climate change through the conservation, restoration, and sustainable management of ecosystems (Colls et al., 2009). Researchers have highlighted the need for reflexive insights, including lessons and challenges implementing EBA projects, given the increased attention it has received in global and national climate discourse (Doswald et al., 2014). Key benefits of EBA have been identified including: (1) securing water resources to help communities cope with drought (2) food and fisheries provision; and (3) buffering people form natural hazards, erosion, and flooding (Munang et al., 2013).

Therefore, this paper presents local EBA examples that demonstrate how Pacific Island communities are leading the implementation of sustainable climate solutions and reinforcing the critical role of ecosystems in climate adaptation. We include examples that address the primary benefits of EBA including water security, food security, and coastal protection. We present examples of EBA projects that were implemented across Micronesia and Melanesia from 2015 to 2018. The EBA project s included a partnership among communities, local governments, and conservation NGOs (The Nature Conservancy, The Micronesia Conservation Trust, other local conservation partners across the Pacific). We discuss these EBA activities, identify barriers to implementation, and highlight the importance of supportive national policies and political will to reinforce and scale up these efforts.

REVITALIZING TRADITIONAL WELLS

Oneisomw (formerly Oneisom) is an island located in Chuuk State lagoon in the Federated States of Micronesia. It has a population of 638 inhabitants (2010 Census of Population and Housing) that is already experiencing the impacts of climate change. Villages are primarily located along the shoreline and are affected by coastal flooding during typhoons and high tide events. The communities rely on a combination of water tanks, aquifers, streams, and wells but freshwater security is threatened by drought and saltwater intrusion. Human impacts are also adversely affecting these freshwater sources and the coastal environment (e.g., pollution from dump sites, waste from pig pens, inadequate sanitation systems, erosion from unpaved pathways, solid waste dumping, and sediment runoff from inland clearing). To improve water security and reduce impacts in the coastal environment, Oneisomw residents have rehabilitated traditional water wells by cleaning them, planting vegetation buffer strips around wells and streams to stabilize degraded banks and reduce sedimentation and installing concrete covers over the wells to reduce trash and other pollutants from entering the wells. They also developed agreements with landowners who had wells to allow others to access water during drought. This approach was presented during a national mayor's summit in 2018 and other communities have requested support to implement these actions to improve water security in their municipalities.

Such local actions need to be reinforced by the implementation of state and national water policies that promote watershed management and provide the foundation for the sustainable use and conservation of water resources (e.g., Pohnpei State Water Policy passed in 2018). This need was articulated at a stakeholder workshop in Pohnpei in 2017 that brought together local leaders, land-owners, and others who utilize the watershed area. While traditional leaders endorsed the process of managing the watershed sustainably, lack of cooperation and planning was noted along with the need to integrate State water management regulations into a national water policy framework to ensure a consistent flow of funds to manage the watershed and protect the full suite of ecosystem services.

IMPLEMENTING CLIMATE SMART AGRICULTURE

Climate-smart agriculture (CSA) is defined as an integrated approach to managing cropland, livestock, forests and fisheries that aims to support food security under the new realities of climate change through sustainable and equitable transitions for agricultural systems and livelihoods across scales (Lipper et al., 2014). It is designed to increase productivity (i.e., produce more food and boost local incomes), enhance the ability of communities to adapt to climate change and weather extremes, and decrease greenhouse gas (GHG) emissions from food production (Steenwerth et al., 2014). When implemented in an island context, CSA can also support benefits to coastal ecosystem (e.g., by reducing sediment into the coastal zone through taro swamps, reducing pressure on wild-caught fisheries, reducing pollutants from fertilizers; Clarke and Thaman, 1993; International Fund for Agricultural Development [IFAD], 2017).

Communities across the Pacific are revitalizing traditional farming practices, based on agroforestry, to increase food security and reduce vulnerability to climate impacts, and they are also experimenting with salt and drought-tolerant crops (FAO, 2010; Mcleod et al., 2018). Traditional farming practices include shading crops with palm leaves, maintaining trees around plants to provide shade, composting using seaweed. Some coastal fishing communities (e.g., Ahus, Papua New Guinea) have historically relied on fishing for food security and are now working with local NGOs, women's groups and government agriculture officers to plant household gardens. Ahus is off the coast of Manus Island in Papua New Guinea and has a population of more than 700 residents. Observed climate impacts include sea-level rise, reduced marine protein sources, saltwater inundation of water wells, coastal erosion, storm surges, droughts, heavy rains, ocean acidification and coral bleaching. With support from the government and NGOs, Ahus has introduced new farming practices that are designed to improve food security, the health of the marine environment, and provide an important source of income for local households (Tara, 2018). These include the introduction of growing food crops including greens, tomatoes and cabbages, composting in very sandy soils, raised gardens and local water collection in drums and small tanks. Women's groups, in partnership with local conservation NGOs and agricultural extension officers have led trainings on farming methods such as the use of organic fertilizers and pesticides, raised beds to improve soil quality and eliminate saltwater intrusion, and the diversification of crops. These farming practices are being replicated and scaled through the provincial women's network Pihi Environment and Development Forum (PEDF). Benefits have included changing and improving the diet of Ahus families, increased cash income for women selling produce at market and to local restaurants, food security especially when bad weather prevents fishing, better community cohesion as people shared ideas and produce.

Low cost aquaculture projects are also being implemented in Ahus, such as clam farming techniques from Palau that have been adapted to local conditions to provide food security and reseed local reefs with clam larvae to re-establish the local wild population. Community members in Tamil, Yap built a nursery utilizing traditional composting techniques and including food crops and plants to revegetate coastal areas vulnerable to erosion (e.g., Nipa Palm). The nursery reduces reliance on coastal fisheries that are being depleted, increases the diversity of food sources, improving community health, and reduces the impact of coastal erosion.

IMPLEMENTATION OF PROTECTED AREAS

Tamil is a municipality on the island of Yap in the Federated States of Micronesia. It includes twelve villages with a total population of about 1200 people living in 848 households (Office of Statistics, Budget and Economic Management, Overseas Development Assistance, and Compact Management, 2011). The community has experienced flooding, erosion, and drought driven by climate change, in addition to saltwater intrusion into freshwater sources and taro patches. Water security is further impacted by poor water management, high dependence on the watershed, and lack of alternative water sources as many local wells are degraded or contaminated by waste and sedimentation from erosion. The community noted the following ecological impacts: declines in coral health, seagrass beds, and reduced fish populations due to increased sedimentation in the coastal environment and pollution run-off driving algal increases (LEAP 2017). To improve water security and coastal ecosystem health, the community declared their first Watershed Protected Area in 2017 (320 acres of watershed protected by traditional council members and recognized by state law). The Tamil watershed provides water to over half of the population of Yap, and its protection provides greater resiliency to and recovery from wildfires, and designates the area as a water conservation zone to increase water security in times of drought.

Similarly, in the island of Chuuk, the community of Oneisomw agreed to implement a locally managed marine area (LMMA) to reduce threats facing coral reefs (e.g., controlling dynamite fishing and overfishing, coral and sand removal, commercial harvesting). The LMMA supports seasonal or permanent closures and fishery management through the traditional management system (*mechen*). Based on the traditional *mechen* system, Oneisomw coral reef "owners" initiated an agreement to collectively enforce seasonal or longer closures of reef areas, based on scientific knowledge and community inputs, to ensure access to coral reef resources for future generations. The LMMA is the first marine protected area for the newly passed Protected Area Network (PAN) legislation. In 2018, the community initiated the process to develop their first land-based protected area by signing a memorandum of understanding with well owners to maintain healthy watersheds. The land-based protected area will reduce pollution and runoff around water sources and will include revegetation with green buffers to help maintain water quality. The next step is for the community to produce a management plan that will integrate a ridge-to-reef approach, which will help to design one of the first Ridge-to-reef protected areas in the country. These collective efforts support the FSM's climate adaptation commitment to the UNFCCC and demonstrate that western and traditional natural resource management methods can be complimentary and mutually beneficial in meeting conservation and human wellbeing goals. They also show how local ideas addressing local needs in the FSM can help to support the ambitious targets of the Paris Agreement.

CLIMATE-SMART DEVELOPMENT PLANS

Melekeok State is located along the east coast of the main island of Palau. The population includes about 300 residents (about 90 households) and the State is also host of the capitol building of the Palau national government. Most of the homes and infrastructure (e.g., elementary school, State office, retirement center) are located along the coast within 5 meters of the high-water mark, thus highly vulnerable to flooding and erosion due to storm impacts and sea-level rise (ADB, 2012; Melekeok State Government, 2012). For example, Typhoon Bopha in 2012 caused significant damage to the community. In response to climate impacts and projections of future impacts, Palau developed a national climate change policy (Government of Palau, 2015) which identifies the need for building ecosystem and community resilience. Additionally, the Melekeok community developed a climate-smart guidance document (Polloi, 2018) due to their high dependence on their terrestrial and marine ecosystems (Brander et al., 2018; Förster, 2018) in partnership with the Melekeok State government and conservation NGOs (e.g., the Nature Conservancy, Micronesia Conservation Trust).

The climate-smart development document provides guidance for updating current infrastructure, designated upland lease development for migrating vulnerable community members and infrastructure away from the coast, and recommendations to make future development less vulnerable to climate impacts. A key focus is to ensure that new development and refinement of existing structures are climate smart and do not cause environmental damages that threaten water quality and the marine ecosystem. For example, the state residential lease/housing program incorporates sustainable designs and approaches to support the resiliency and enhancement of ecosystem services. The residential lease agreement requires individuals to revegetate bare soils to reduce run-off and sedimentation into the coastal system, minimize stormwater flow to promote water infiltration and support water supply, install water catchment systems to reduce vulnerability to drought, and include renewable energy systems (e.g., solar panels) through existing national loan programs. In addition, new permits for land use, the development of residential areas, and commercial developments require measures that support water security and erosion control (e.g., hedge rows and filter strips to mitigate soil erosion). Melekeok State leadership is also considering legislation for climate proofing new residential houses that would require new houses to use hurricane clips in the construction.

These innovations in Palau provide a model for how to develop climate-smart development that also include benefits to the coastal and marine ecosystem. To upscale implementation and enforcement at national level, policies are needed that support sustainable financing mechanisms. Access to loans for building new homes should be provided under the condition of complying with guidance for climate-smart homeowners, similar to the Energy Efficiency Subsidy Program of the National Development Bank of Palau. Such policies could enhance the upscaling of adaptation strategies and their inclusion in local and national infrastructure development programs.

CHALLENGES TO IMPLEMENTING ADAPTATION STRATEGIES

A number of challenges threaten the success of local community-based adaptation projects including the remoteness of some islands, lack of capacity to implement and sustain projects, lack of governance and the way that impact is measured.

Remoteness of Islands

Logistical, technological, and weather-related obstacles are common in remote islands in the Pacific, causing delays to material-dependent projects. High costs of transportation and certain goods divert spending from on-the-ground implementation. Distance from markets can also limit economic growth. Such issues can lead to decreased interest in the region from international conservation supporters and investors. However, the logistical challenges and high costs related to often remote locations of islands is also a factor driving the development of local solutions for climate adaptation that build on local traditional knowledge. While some of the solutions are specific to the needs of islands, they inspire innovative approaches that can be applied in other areas.

Lack of Technical and Financial Capacity

Pacific Island countries face a number of capacity constraints (e.g., financial and project management, climate modeling and spatial analysis, and infrastructure maintenance; Dornan and Newton Cain, 2014). Sustained capacity in the local NGOs also is a challenge; as talented youth rise through the ranks of conservation programs, they are often recruited into higher-paying government or private sector jobs or seek opportunities abroad. Such staff turnover problems hinder long-term conservation projects by causing significant portions of funding sources to be repeatedly used toward capacity development. Local adaptation projects supported by external sources of funding (e.g., climate grants) often end when the grant is over, if there is not sufficient local capacity to continue the project. Finally, lack of technical capacity is also a challenge.

For example, enforcement of marine resource harvesting regulations requires expensive investments in equipment (e.g., boats and surveillance technologies) and advanced training. Enforcement funding is often gleaned from the end of project budgets, as expenditures such as staff time, materials, and planning commonly absorb substantial amounts of initial funding. Technical capacity for climate resilient agriculture is limited, and on-going support is often needed to address emerging threats (e.g., new garden pests in Ahus, Papua New Guinea).

Governance

Complex land tenure structures commonly follow traditional or tribal governance systems which can conflict with Western judicial laws and processes, making governance approaches ineffective. This can deter climate financing from large international organizations who require stringent contract-based agreements such as land transfers and easements for protected areas. Nevertheless, traditional tenure and knowledge systems can inform sustainable adaptation strategies and must be considered in the design of adaptation policies. Hence there is the challenge of ensuring compatibility between traditional and western governance systems. The recently established Local Communities and Indigenous Peoples Platform (LCIPP) under the UNFCCC can help to bridge these institutional challenges and ensure local traditional knowledge is considered in the provision of adaptation finance.

Measuring Impact

Many Pacific islands have small populations and small land masses. If donors prioritize their support based on the total number of hectares protected/restored or the total number of people who benefit from a given intervention, Pacific Island projects may not be selected for funding. However, the strong dependence on island communities on their ecosystems for food, livelihoods and traditional practices, provides opportunities for demonstrating how climate adaptation projects can result in direct benefits to both ecosystems and human wellbeing. Additionally, regional commitments to conservation and sustainability such as the Micronesia Challenge can be an important mechanism to scale conservation efforts by providing enabling conditions to better cope with climate change. Initiated by a coalition of regional governments and endorsed at an international level with sustainable funding and technical support for implementation, the Micronesia Challenge serves as a model for other regions. Indeed, it inspired the development of the Caribbean Challenge, Western Indian Ocean Challenge, and the Coral Triangle Initiative.

SCALING ECOSYSTEM-BASED ADAPTATION THROUGH SUPPORTIVE NATIONAL POLICIES AND INNOVATIVE FINANCING

Ecosystem-based adaptation actions that support human wellbeing and healthy ecosystems require financing and supportive policies to ensure their implementation, sustainability, and scaling across the region. Such policies must be continually evaluated and refined to ensure that they continue to address local needs in response to change social, ecological, and climatic conditions and must be developed in concert with traditional knowledge. For example, marine protected areas in Manus, Papua New Guinea work best when they reflect the latest science on fish movements and aggregation sites and also follow local tribal boundaries to enable clans to manage their customary land and seas as part of the protected area. This means that local tribes set the rules for their marine protected area that enable species sustainable and address local needs. Thus, in some communities (e.g., Ahus, Papua New Guinea), it is important to strengthen tribal governance and local institutions to mobilize resources and management of adaptation projects. Methods to do so include incorporating climate change into existing ward plans, aligning ward plans with existing provincial and government policies and plans and adapting these plans over time to address changing conditions.

Learning exchange between local, state and national governments are an important mechanism to discuss the challenges communities are encountering in adapting to climate change and to refine current policies with new scientific and local knowledge. They also can highlight gendered impacts of climate change and the differential capacities for adaptation. For example, women in some Pacific Islands are not entitled to land rights due to customary laws and practices which may limit their ability to grow food and resettle in areas less vulnerable to climate impacts. Therefore, policies are needed that consider these gendered impacts (e.g., addressing land ownership inequity as climate change is reducing the available land in some places such as Papua New Guinea; Mcleod et al., 2018).

Innovative financing for ecosystem-based adaptation includes the development of tools (e.g., green fees, payment for ecosystem services) and new partnerships with the private sector. For example, water utilities and other businesses that utilize nature for profit can be incentivized to protect the environment. Utilizing payment schemes, such as payments for ecosystem services, creates financial mechanisms to ensure that water is clean, sustainable, and generates new sources of revenue for watershed protection.

CONCLUSION

The examples above demonstrate positive steps taken by local communities and partners to implement EBA projects in small islands states, yet there is little systematic information on the large-scale effects of these measures for building climate resilience across the region. While some island communities can build resilience to climate change, others will face the limits of adaptation and use migration as a last resort for adapting to climate change impacts. Assessments that identify and predict where adaptation limits are likely to occur and who is most likely to be affected are essential to better plan for climate impacts (Dow et al., 2013). Further, scientific assessments that provide evidence for the effectiveness of the EBA projects are lacking, especially those that include controls to assess the impacts of interventions and provide plausible counterfactual arguments regarding causal mechanisms (Reid, 2011; Munroe et al., 2012). Research is also needed to highlight social, ecological, and economic opportunities for upscaling ecosystem-based adaptation and to assess the contribution of adaptation to enhancing island resilience to climate change. Current assessments tend to focus on quantifying biophysical and socio-economic benefits but fail to make the link to management and policy options that enable the implementation of local adaptation options (Hills et al., 2013). In addition to research needs, there is the need for combining traditional with more recently introduced governance systems. Cross-regional exchanges and capacity building can foster the development of innovations that tackle the challenge of including local traditional knowledge and address the needs of island communities. Furthermore, platforms and partnerships that bring together leaders of traditional governance systems with representatives of Western governance systems can help to overcome barriers between different institutional systems and encourage the implementation of holistic community- and ecosystem-based adaptation approaches.

AUTHOR CONTRIBUTIONS

EM conceived of and developed the manuscript with contributions from MB-A, JF, CF, BG, RJ, GP-K, MT, and ET. JF, CF, GG, BG, RJ, GP-K, MT, and ET collected the data that supported the analysis.

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Developing a Social–Ecological–Environmental System Framework to Address Climate Change Impacts in the North Pacific

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"Forecasting and Understanding Trends, Uncertainty and Responses of North Pacific Marine Ecosystems" (FUTURE) is the flagship integrative Scientific Program undertaken by the member nations and affiliates of the North Pacific Marine Science Organization (PICES). A principal goal of FUTURE is to develop a framework for investigating interactions across disciplinary dimensions in order to most effectively understand large-scale ecosystem changes and resulting impacts on coastal communities. These interactions are complex, often nonlinear, occur across a range of spatial and temporal scales, and can complicate management approaches to shared and *trans*-boundary problems. Here, we present a Social–Ecological–Environmental Systems (SEES) framework to coordinate and integrate marine science within PICES. We demonstrate the application of this framework by applying it to four "crisis" case studies: (a) species alternation in the western North Pacific; (b) ecosystem impacts of an extreme heat wave in the eastern North Pacific; (c) jellyfish blooms in the western North Pacific; and (d) Pacific basin-scale warming and species distributional shifts. Our approach fosters a common transdisciplinary language and knowledge base across diverse expertise, providing the basis for developing better integrated end-to-end models. PICES provides the structure required to address these and other multi-national, inter-disciplinary issues we face in the North Pacific. An effective and comprehensive SEES approach is broadly applicable to understanding and maintaining resilient marine ecosystems within a changing climate.

Keywords: North Pacific, North Pacific Marine Science Organization, social–ecological systems, climate change, ocean sustainability

CHALLENGES IN CHARACTERIZING CHANGES IN THE NORTH PACIFIC

Long-term observations of physical and biological properties collected around the North Pacific, coupled with numerical simulations of coupled atmosphere-ocean-ecosystem phenomena, have improved understanding of the drivers of climate variability in the North Pacific and the consequent impacts on marine ecosystems. This body of research has highlighted patterns of climate variability associated with El Niño-Southern Oscillation (ENSO) at interannual scales (e.g., Doney et al., 2012) and the Pacific Decadal and the North Pacific Gyre Oscillations (PDO and NPGO, respectively; Mantua et al., 1997; Di Lorenzo et al., 2008) at decadal to multi-decadal scales. The influence of long-term anthropogenic climate change on the North Pacific basin is increasingly evident (Barange et al., 2016; Holsman et al., 2018). Further, coupled numerical models and continued observations have improved our understanding of the feedbacks and teleconnections among tropical ENSO events and PDO and NPGO patterns in the extratropical North Pacific (Di Lorenzo et al., 2013; Newman et al., 2016), and the sensitivity of such dynamics to anthropogenic climate change continues to stimulate new questions. The desire to understand the basin-scale climate and ocean dynamics, to predict variability in ocean conditions and the consequences of those processes for marine ecosystems and human society, and to communicate scientific understanding to decision makers and the public motivated the development of the intergovernmental North Pacific Marine Science Organization (PICES).

Our understanding of the principal drivers of large-scale climate variability in the North Pacific is quite mature (Box 1; Liu and Di Lorenzo, 2018). However, the mechanisms by which that variability impacts marine ecosystems, at both regional and basin scales and across multiple trophic levels, remains poorly understood. Furthermore, the ways in which human societies respond to these ecosystem fluctuations can be complex and inconsistent, depending on varying regional and national motives and contemporary concerns (Ommer et al., 2011). Interactions among social–ecological systems (SES; Berkes and Folke, 1998), occur across a range of spatial and temporal scales, contributing to the challenges in studying and managing these systems. In the face of a large-scale global driver like climate change, there is a community-wide goal to maintain resilient and sustainable ecosystems, requiring a more complete understanding of climate-driven impacts on marine ecosystems that can inform effective strategies of marine management and governance.

Here, we synthesize recent developments in understanding climate variability in the North Pacific, its ecosystem impacts, and how human societies affect, and are affected by, these environmental and ecological changes. Building from the SES framework, we review the concept of social–environmental–ecological systems (SEES), and describe how a SEES approach has been implemented within the PICES, through its flagship Science Program "Forecasting and Understanding Trends, Uncertainty and Responses of North Pacific Marine Ecosystems" (FUTURE)[1]. To illustrate how PICES addresses complex, multi-dimensional and multi-national issues in the North Pacific, we apply the SEES approach to four case studies in which specific climate drivers have resulted in ecosystem perturbations and responses within human societies. Finally, we review the lessons learned from PICES' approach to understanding climate-ecosystem-human interactions, and identify the key challenges remaining.

The goal of the PICES FUTURE Program is to "understand and forecast the responses of North Pacific marine ecosystems to both climate change and human activities, and to evaluate the capacity and resilience of these ecosystems to withstand perturbations" (PICES, 2016). Specifically, the principal objectives of FUTURE are to:

(1) Increase understanding of climatic and anthropogenic impacts and consequences on marine ecosystems, with continued leadership at the frontiers of marine science;
(2) Develop activities that include the interpretation, clarity of presentation, peer review, dissemination, and evaluation of ecosystem products (e.g., status reports, outlooks, forecasts).

To address objective (1), PICES has outlined a series of research questions (**Appendix 1**) that guide the work of Expert Groups (Sections, Working Groups, Study Groups and Advisory Panels)[2]. To address objective (2), PICES produces a number of products[3] aimed at communicating PICES science to a diverse audience, including the scientific community, marine resource management agencies within the member countries, other international marine science and management organizations (e.g., ICES, RFMOs), and the general public. Given its objectives and legacy of multi-disciplinary research on the North Pacific, PICES, and the FUTURE program in particular, are ideal candidates to explore the many changes taking place in the North Pacific within a SEES approach.

THE NORTH PACIFIC SOCIAL–ECOLOGICAL–ENVIRONMENTAL SYSTEM (SEES)

Largely as a result of the separation of ecological and social sciences in resource management issues (e.g., fisheries), natural and human systems have usually been considered as two separate entities in the marine realm (Berkes, 2011). In this concept, natural systems formed the template within which human systems operated (e.g., Park, 1936), and human systems were seen as drivers and recipients of change from the natural system. About 20 years ago, however, this view began to shift (driven by natural resource challenges and the inability of the previous model to provide lasting, meaningful solutions, e.g., Berkes and Folke, 1998)

[1] https://meetings.pices.int/Members/Scientific-Programs/FUTURE
[2] http://meetings.pices.int/about/OrganizationStructure
[3] http://meetings.pices.int/publications

toward a concept of a fully coupled and interacting social–ecological system (Perry et al., 2010). Berkes (2011) notes that this concept recognizes the social (human) and ecological (biophysical) subsystems as two equally important parts, which function as a coupled, interdependent, and co-evolutionary system. As described by Berkes (2011, p. 12), "Human actions affect biophysical systems, biophysical factors affect human well-being, and humans in turn respond to these factors".

As the social–ecological system concept has evolved, it has expanded beyond its original common-pool resource management (mostly fisheries) origins. McGinnis and Ostrom (2014) describe how the initial focus involved resource users who extracted units from a resource system, and how these users maintained an overarching governance in the context of related ecological systems and the broader social, political, and economic setting. McGinnis and Ostrom (2014) further propose an expanded social–ecological system framework to guide analysts in many disciplines with studying similar sets of problems. This relies on addressing three questions:

(1) What is the focal level of analysis (e.g., what system, which actors, what governance regime)?
(2) What variables should be measured and how can indicators for these variables be developed and implemented?
(3) How can the results be communicated across diverse research (and management) communities?

PICES has further elaborated these three questions of this social–ecological systems concept to address a variety of coupled marine social and ecological changes in the North Pacific. In particular, PICES has explicitly identified the climate system and its effects on the physico–chemical ocean environment, as necessary to fully understand current changes taking place within the North Pacific, and expressed this as a coupled social–ecological–environmental system (SEES).

The PICES implementation of a North Pacific SEES was designed to identify and understand the linkages between climate forcing, oceanic processes, marine ecosystem responses (at multiple trophic levels and spatial scales), and the human system (**Figure 1**). Within the climate system, we aim to understand the modes of climate variability and change on the basin scale, and how this climate forcing downscales to the coastal domain and to regional scales, which are relevant for the management of marine resources. These climate drivers subsequently impact physical, chemical, and biological processes across a range of spatial and temporal scales, which can collectively affect the functioning and ultimately resilience of marine ecosystems. Climate-driven impacts on the ecosystem can occur at all trophic levels and can alter habitats, species-specific functional responses and population and community structure. At the ecosystem level, these changes can alter overall resilience and shift key thresholds (i.e., tipping points) and ecosystem reference points that are critical for effective management. Thus human societies, which rely heavily on the ecosystem services provided by the ocean, are often negatively impacted by these ecosystem fluctuations. Human activities (e.g., fisheries and aquaculture, shipping) also contribute multiple stressors back onto the ocean and its ecosystems (e.g., harmful algal blooms, invasive species, noise and pollution), thereby linking the human system back to the environmental and ecological dimensions. Finally, to better understand how each of the SEES dimensions varies and interacts requires adequate monitoring and assessment of all of its components, and the subsequent dissemination of data and products.

Within PICES, this SEES approach for the North Pacific accomplishes three goals: (a) it provides a roadmap for initiating interactions amongst PICES Expert Groups and for developing products to fulfill FUTURE's objectives; (b) it identifies critical knowledge gaps that PICES might address through the creation of new Expert Groups; and (c) it facilitates a holistic understanding of how large-scale climate variability and change impacts oceanic and ecosystem processes, and how human societies can manage, mitigate, benefit from, and/or adapt to these changes.

UNDERSTANDING AND SOLUTIONS THROUGH A SEES APPROACH: CASE STUDIES

The SEES approach relies on "embracing reciprocal links among people and nature, and harnessing knowledge from the natural and social sciences" (H. Leslie, pers. comm.; Leslie, 2018). In the North Pacific, PICES has implemented a SEES framework to facilitate bridging scales between local communities and basin scale dynamics, and to better understand complex dynamics that impact its coastal communities (i.e., within member nations). To demonstrate the robustness and utility of this approach, we applied the SEES concept to four "crisis" case studies in the North Pacific.

Case Study 1: Species Alternation in the Western Pacific

Fish species alternation is a classic example of an ecosystem regime shift response to climate change (Alheit and Bakun, 2010). Such an alternation occurred in the western North Pacific following the climate regime shift of 1988–1989 (Zhang et al., 2007; **Figure 2**). The Japanese sardine (*Sardinops melanostictus*) stock was at historic highs (>20 million tons) in the mid-1980s and showed a rapid and continuous decline after 1988 to a level of<1 million tons by the mid-1990s. In the winter of 1982–1983, an altered wind field in the eastern North Pacific induced positive sea surface height anomalies (+SSHA) in the region, which subsequently propagated westward as a Rossby wave (Nonaka et al., 2006) ①. The +SSHA reached the Kuroshio Extension region in 1988, resulting in increased wintertime sea surface temperatures (SST) as the mixed layer depth shoaled (Nishikawa and Yasuda, 2008) ②. These changes in the physical properties in the Kuroshio Extension, the nursery ground of larval and juvenile Japanese sardine, subsequently reduced and changed the timing of regional primary production. The timing of the spring bloom after 1988 was up to 2 months earlier

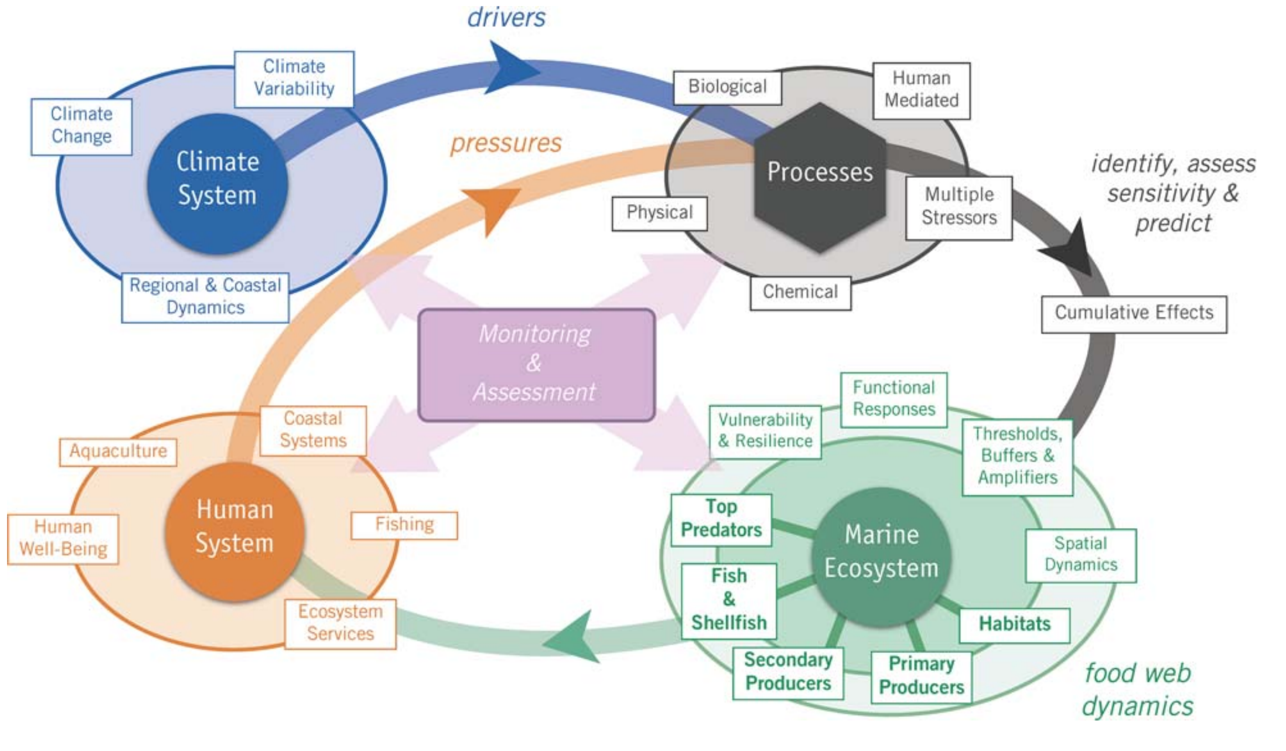

FIGURE 1 | The North Pacific Social–Ecological–Environmental System (FUTURE schematic).

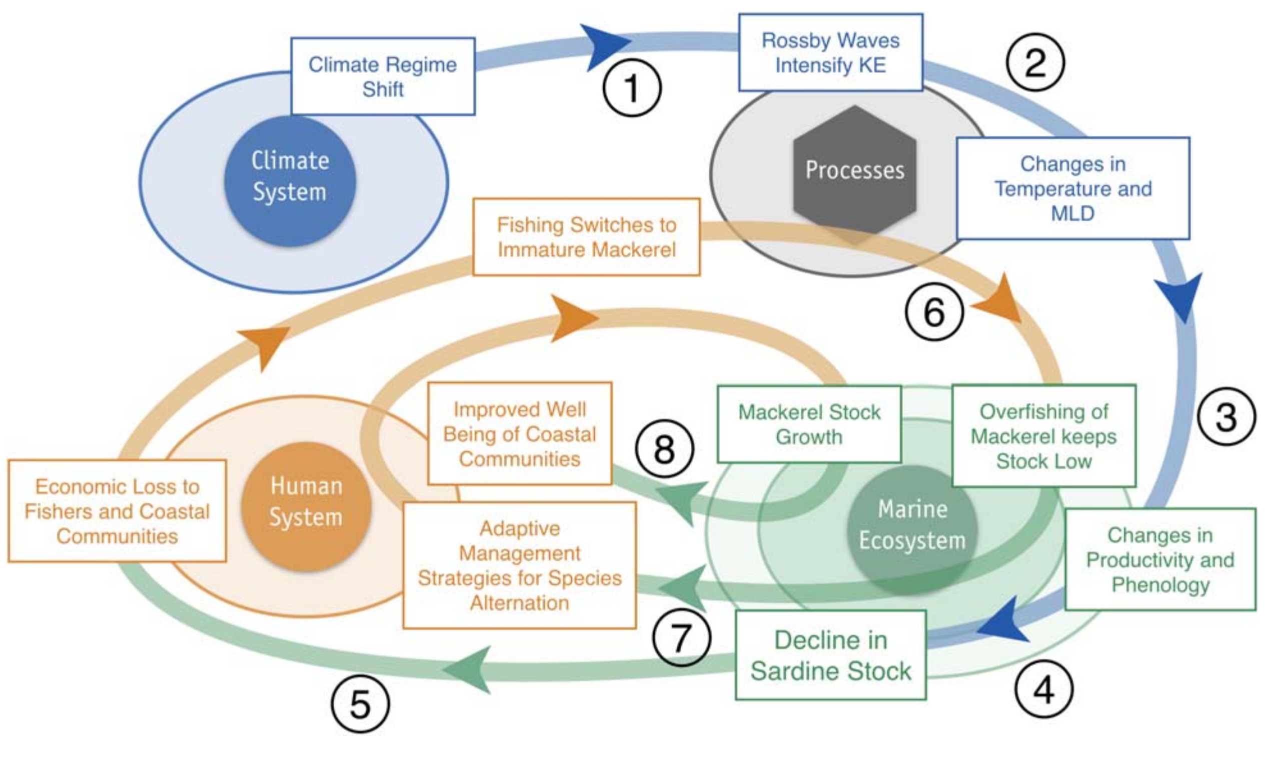

FIGURE 2 | FUTURE SEES case study 1: Species alternation in the western Pacific.

than before (Nishikawa et al., 2011) ③. As a result, Japanese sardine recruitment decreased due to a mismatch between the peak of prey production (February) and the arrival of larval and juvenile sardine (April) ④. By the mid-1990s the stock had collapsed.

Purse seiners and local communities suffered economic losses from the collapse of the Japanese sardine stock ⑤. Most purse seiners had invested in large-scale vessels through the late 1980s, and were unable to pay off loans on those vessels when the sardine catch declined. Local communities also suffered economically because of the small amount of sardine landings used as raw materials for processing. To avoid bankruptcy, purse seiners switched their target catch to immature chub mackerel (*Scomber japonicus*), eventually leading to the overfishing of this stock in the 1990s (Makino, 2011) ⑥. Although there were several strong year classes of chub mackerel during this period, overfishing occurred and resulted in recruitment failure of the spawning population (3 + years) and a drop to low stock levels.

To address these issues, the national government of Japan introduced a "total allowable catch" in 1997, setting an upper limit on chub mackerel total catch. In addition, to protect the strong year classes, the government and purse seiners cooperatively introduced the "Resource Recovery Plan" in 2003 which adaptively controls fishing pressure on immature chub mackerel when a strong year class occurs (Makino, 2018) ⑦. Since adoption of this plan, several strong year classes have been protected thereby allowing the chub mackerel stock to increase to sustainable levels (Yukami et al., 2017). Overall, the economic situation of these Japanese purse seiners and the well-being of their coastal communities have been improved ⑧.

Since these fish species alternations were induced by natural climate variability, it is expected that the regional chub mackerel stock may decline in the future, being replaced by an increasing sardine stock. However, an increased understanding of the dynamics associated with these alternations, from climate regime shifts to fisher behavior and the effects of both governmental and industry interventions, provides an important basis for understanding future changes. Continued monitoring of the physical (environmental) conditions, plankton production and phenology, and larval fish survival in this region will be essential to identify ecosystem change and inform adaptive management strategies for coastal fishers.

Case Study 2: Ecosystem Impact of a Marine Heat Wave in the Eastern Pacific

The northeast Pacific Ocean experienced highly anomalous atmospheric and oceanic conditions during 2014–2016, which was accompanied by significant ecosystem disruptions along the North American West Coast (**Figure 3**). A large warm temperature anomaly (nicknamed "The Blob") developed in the upper ocean during fall 2013, and spread through much of the Gulf of Alaska during the winter of 2013–2014, reaching record-breaking SST anomalies (> 3 SD; Bond et al., 2015; Di Lorenzo and Mantua, 2016; Hobday et al., 2018) ①. The anomaly persisted through the winters of 2014–2015 and 2015–2016, with warm SST anomalies reaching the west coast of North America in spring and summer 2014 and extending from Alaska to Baja California by spring 2015 (Kintisch, 2015; Di Lorenzo and Mantua, 2016).

As these warm near-surface waters were advected to and impacted coastal waters, the enhanced vertical stratification reduced the efficacy of coastal upwelling to supply nutrients to the euphotic zone which negatively impacted coastal productivity ②. The combination of reduced primary productivity and the presence and persistence of unusually warm waters led to significant disruptions in the California Current ecosystem (Cavole et al., 2016), including reduced phytoplankton abundance and production (Du and Peterson, 2018; Gómez-Ocampo et al., 2018), a coastwide toxic algal bloom (McCabe et al., 2016; Ryan et al., 2017), reduced biomass of copepods and euphausiids and high abundance of oligotrophic doliolids (Peterson et al., 2017), the massive mortality of a planktivorous seabird (Jones et al., 2018), and substantial changes in species distributions and community composition across multiple trophic levels (Cavole et al., 2016; Santora et al., 2017; Brodeur et al., 2019) ③.

These ecosystem disruptions had immediate and profound impacts on the human communities that rely on the marine resources of the California Current. The harmful algal bloom led to the closure of lucrative salmon fisheries and changes in the timing of crab fisheries (Cavole et al., 2016) ④. These changes led to disproportionate negative impacts on small-scale fishers and subsequent economic loss to their coastal communities (McCabe et al., 2016) ⑤. The anomalous conditions led to a particularly unfortunate convergence of circumstances leading to higher whale mortality. While the HAB event delayed the opening of the crab fishery, the anomalously warm conditions within the California Current led to a higher proportion of anchovies, a key forage fish, to inhabit nearshore regions where crab pots are typically deployed. Humpback whales, which migrate through the region, foraged further inshore for anchovies just as the crab fishery was opened, leading to a higher number of whale entanglements and mortalities (NOAA Fisheries, 2017) ⑥.

There were important management actions taken to respond to this unprecedented situation. Fishers made requests for a disaster declaration (McCabe et al., 2016), and both State and Federal agencies set up committees of managers, scientists, fishers and NGO representatives to develop adaptive management strategies ⑦. There were changes in marine spatial planning, and an NGO provided funds to the fishing community to pay for removal of derelict fishing gear ⑧. Although the ecosystem response to this large marine heat wave was unanticipated, the human response was relatively quick and likely mitigated some of the more significant negative impacts. Extreme events such as this are projected to become more frequent with climate change (Sydeman et al., 2013; Froelicher et al., 2018), suggesting that a SEES approach such as that applied here can confer resiliency to the human communities that depend on the sea. In particular, monitoring environmental and ecosystem

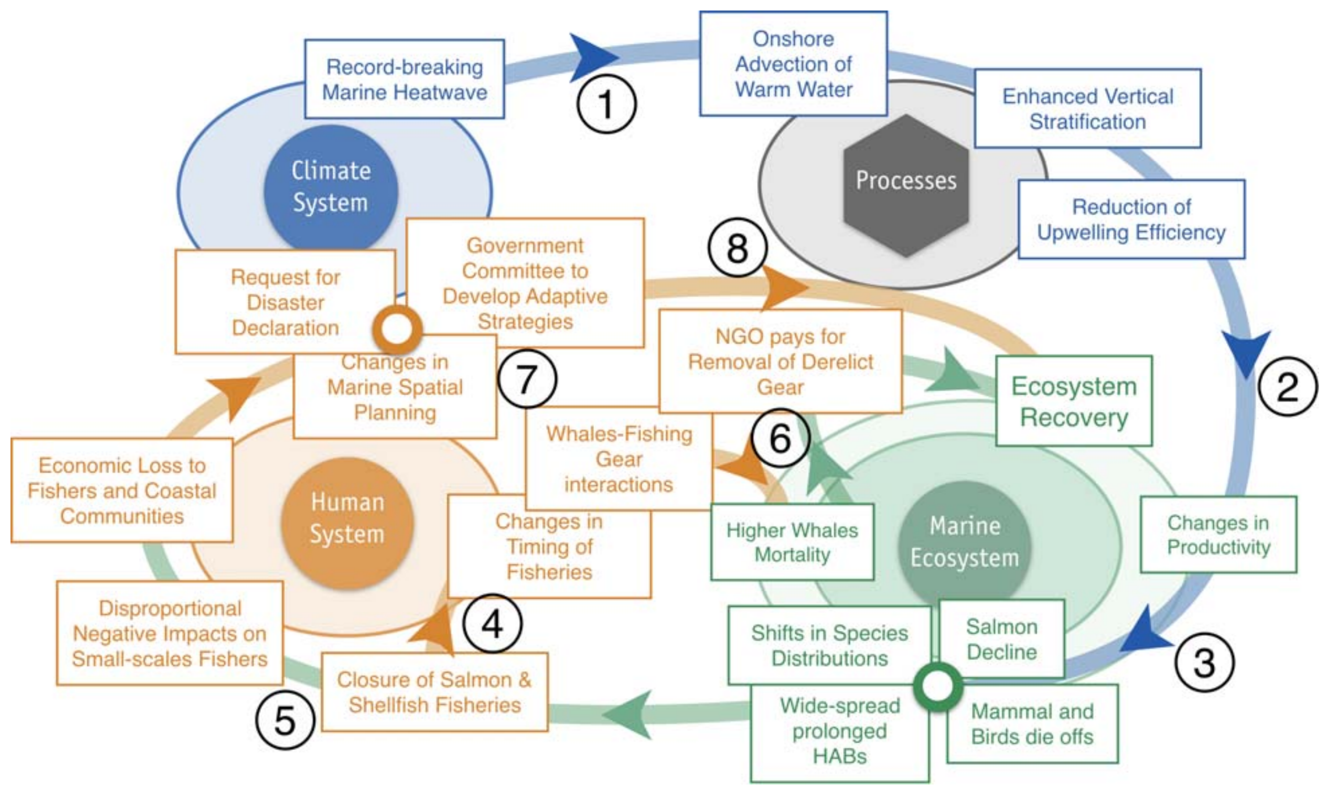

FIGURE 3 | FUTURE SEES case study 2: Ecosystem impact of a marine heat wave in the eastern Pacific.

conditions at sufficient spatial and temporal resolution, as well as human interactions with the ecosystem, will allow relevant stakeholders to respond more efficiently to large-scale perturbations such as these.

Case Study 3: Jellyfish Blooms in the Western Pacific

Coastal marine ecosystems are exposed to multiple anthropogenic stressors that can degrade the ecosystem services in unexpected ways. One such example is large-scale jellyfish blooms which cause economic losses to fishers and coastal communities. These blooms often decrease fish stock biomass, value, and marine recreation while increasing the costs associated with preventing clogging of cooling pipes, including power generating facilities (Uye and Brodeur, 2017). A number of human activities related to coastal development can promote the survival of jellyfish in their early life stages, particularly with newly developed platforms and coastal infrastructure providing more substrate (habitat) for polyp settlement and survival (Duarte et al., 2013; Makabe et al., 2014; **Figure 4**) ①. Eutrophication allows for higher abundances of microzooplankton which are prey for both benthic polyps and planktonic ephyrae, and the resultant hypoxia eliminates predators of polyps while the polyps themselves can tolerate these low oxygen conditions ②. Fishing pressure (pathway #3 in **Figure 4**) can also eliminate predators of ephyrae and small medusae (Shoji, 2008) ③. Finally, a winter warming trend observed in the western North Pacific can accelerate asexual reproduction and polyp growth (Han and Uye, 2010) ④.

These anthropogenically driven environmental changes collectively contribute to an increase in the abundance of jellyfish (Purcell et al., 2007; Richardson et al., 2009). Jellyfish feed on fish larvae and mesozooplankton, which are an important prey for fish and a predator of microzooplankton, so subsequent declines in fish are beneficial for the survival of polyps, ephyrae and medusae – a positive feedback referred to as the "jellyfish spiral" (Uye, 2011). Large jellyfish blooms have occurred more frequently in the western North Pacific in recent years, affecting coastal activities in a number of countries including Japan and the Republic of Korea (**Figure 4**). Blooms of the giant jellyfish *Nemopilema nomurai* occurred frequently after 2000 in the marginal seas of the western North Pacific (Uye, 2008), resulting in substantial economic losses to fishers and coastal communities ⑤. The impact was especially serious for coastal fishers because the presence of giant jellyfish impeded the catch of commercially valuable fish species and decreased fish prices due to reduced catch quality. In response to these blooms, a collaborative international monitoring program was established (Uye and Brodeur, 2017). This program has allowed the size of jellyfish blooms, and their dispersal by ocean currents, to be monitored (Xu et al., 2013; Sun et al., 2015) ⑥. Based on these observations and model simulations, it is now possible to provide early warnings of jellyfish blooms (magnitude, timing) to the fishing community and other stakeholders

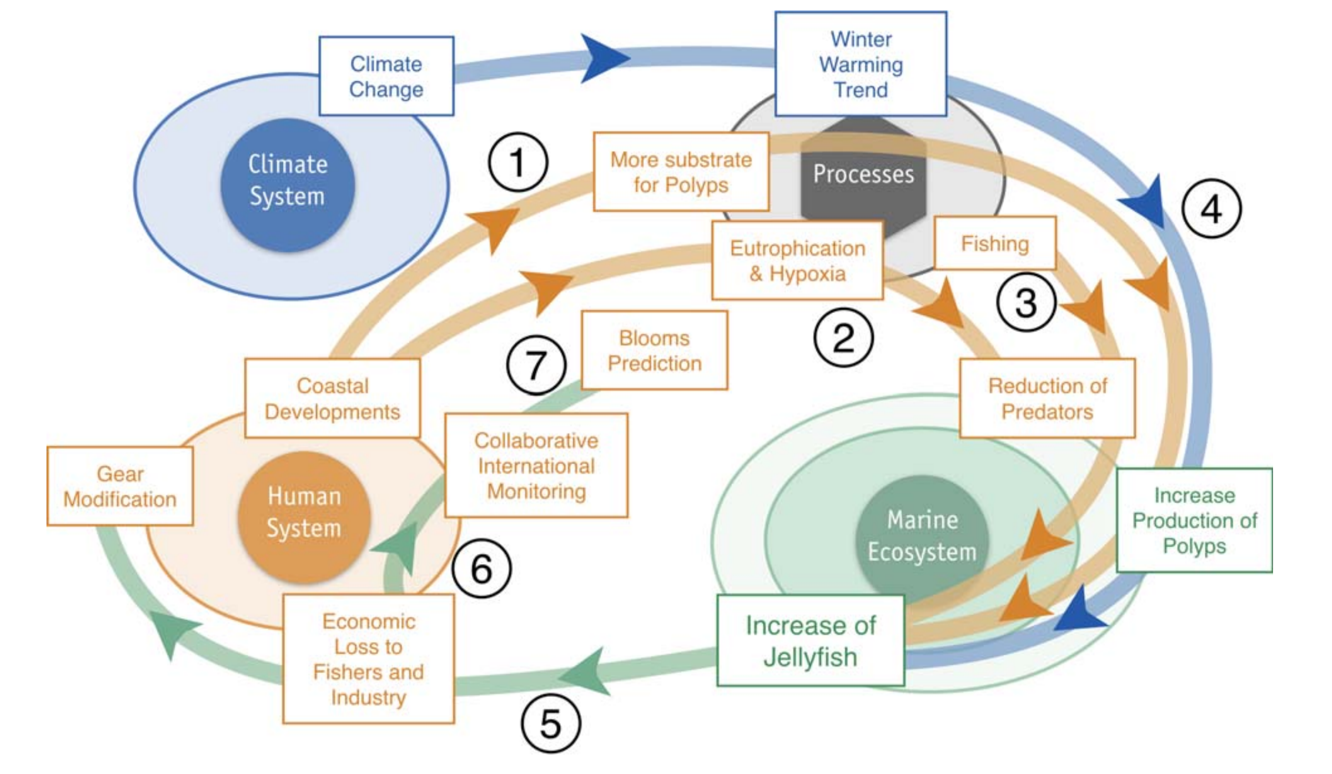

FIGURE 4 | FUTURE SEES case study 3: Jellyfish blooms in the western Pacific.

(Uye, 2014) ⑦. These forecasts are based on early-summer on-deck sightings of young medusae from ferries, allowing fishermen to prepare countermeasures in anticipation of potential jellyfish blooms.

Case Study 4: Warming and Distributional Shifts in Highly Migratory Species

Climate models project a warming of sea surface temperatures of up to 3°C in the North Pacific by the end of the 21st century (Woodworth-Jefcoats et al., 2017) ①. In response to this basin-scale warming, many pelagic fish species are expected to shift their migratory patterns and distributions, both due to physiological constraints and changes in prey distribution (Cheung et al., 2015; Woodworth-Jefcoats et al., 2017; **Figure 5**) ②. In particular, species such as bluefin, yellowfin and albacore tuna are likely to shift their centers of distribution poleward to maintain an optimal thermal environment and to follow shifting prey populations (Hazen et al., 2013) ③. Indeed, during the recent extreme marine heat wave in the eastern North Pacific in 2014–2015, bluefin tuna were observed well north of their typical range, fostering robust recreational fishing opportunities off northern California and Oregon that extended into Washington and British Columbia (Cavole et al., 2016).

These distributional shifts, especially of top predators, will lead to profound changes in the abundance (and possibly distribution) of commercially important fish species that are available to local fishing communities ④, potentially leading to loss of revenue and disruptions in the efficacy of fisheries management protocols (e.g., Pinsky and Fogarty, 2012). Fishing communities that are reliant on particular stocks may face substantial hurdles, as fishing trips increase in distance and expense, and may even cross jurisdictional boundaries. Further, as new species begin to inhabit local waters, they could face over-exploitation if there are insufficient management policies in place to account for the new ecosystem structure. Regardless of how the ecosystem is altered, there will be winners and losers, both in terms of the dominance of particular species and the fishing communities that may or may not be able to adequately respond to these changes (e.g., McIlgorm et al., 2010; Perry et al., 2011) ⑤.

These climate-induced changes in species distribution (e.g., Humboldt squid in the Northeast Pacific; Stewart et al., 2014) raise important policy and management issues. Within the coastal boundaries of the North Pacific, negotiations will be required to sustainably manage *trans*-boundary stocks, especially for emerging *trans*-boundary species. For example, there are negotiations underway between Canada, The United States and Mexico to adapt existing policy and management options to shifting sardine distributions. Similar negotiations are underway between Canada and The United States to consider the poleward shift of albacore tuna populations. In waters beyond national jurisdictions, Regional Fishery Management Organizations (RFMOs, such as the recently established North Pacific Fisheries Commission) will need to account for projected distributional shifts as new policy, regulations and management considerations

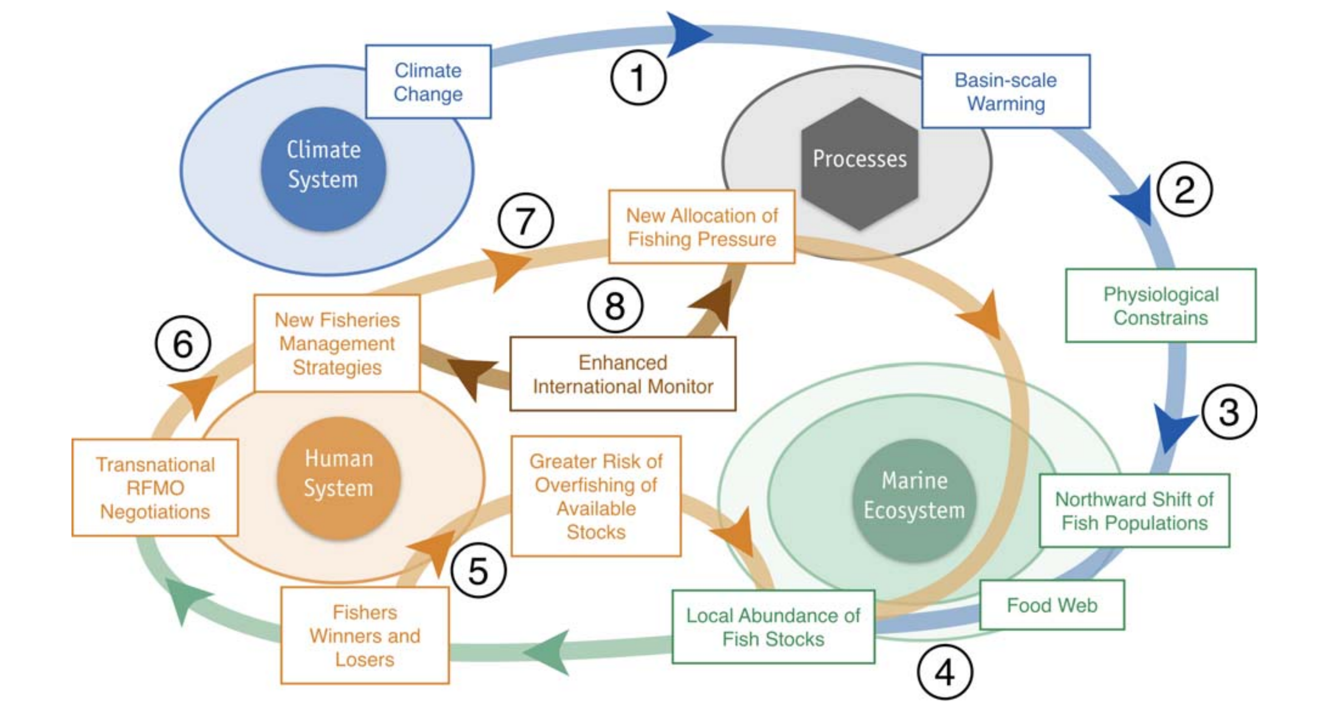

FIGURE 5 | FUTURE SEES case study 4: Warming and distributional shifts in highly migratory species.

are developed ⑥. These new climate-informed management strategies will open new fishing opportunities to nations that fish the North Pacific ⑦. Any efforts to adaptively respond to distributional shifts will also require enhanced monitoring and assessment of the ecosystem, both regionally and basin-wide, which in turn will require international cooperation ⑧. The SEES framework applied here could be instrumental in understanding and forecasting potential interconnections between social and ecological systems.

LESSONS LEARNED AND NEW CHALLENGES

We have implemented a Social–Ecological–Environmental framework to address critical issues of relevance to nations that share North Pacific marine resources, specifically the member nations of PICES, with a focus on climate- and human-induced ecosystem changes that impact coastal communities. This approach has increased capacity for PICES to understand and communicate the processes that link climate variability and change to multi-trophic, multi-scale ecosystem responses, and to more effectively develop strategies to mitigate negative impacts on both our ecosystems and the human communities that depend upon them. PICES has used this approach to identify key linkages (between individual scientists, national and international organizations, and research projects) for enhanced collaborative research, as well as to identify important gaps in research and communication that require attention. Within PICES, our SEES approach has led to the creation of several new multi-national Working Groups that are addressing issues of particular concern, including one comparing thresholds of ecosystem responses within national Exclusive Economic Zones and another aimed at improving short- (seasonal) to long-term (decadal) ecological forecasting[4] on both coastal and basin-wide scales.

We demonstrated our SEES approach by describing the cross-disciplinary linkages associated with four important issues affecting PICES member nations (**Figures 2–5**). Working through these examples has allowed PICES to better address these issues by strengthening communication pathways and focusing limited resources on shared problems, and paves the way for developing end-to-end (physics to humans) models of the system. Although we chose these case studies based on our collective knowledge of the issues, the approach could also be applied in anticipation of other climate-induced impacts (e.g., effects of increased ocean acidification or declining oxygen levels) as well as anthropogenically driven stressor-response cases beyond those associated with climate (e.g., an oil spill, coastal development, etc.). This approach is broadly applicable to other inter-governmental organizations whose mandate is to address issues that transcend national and traditional disciplinary boundaries. A key challenge will be to develop effective means to translate the products of our SEES framework – an improved understanding of the linkages between the climate system, the marine ecosystem, and human communities – to the managers and stakeholders tasked with preparing society for the forthcoming changes.

[4]https://meetings.pices.int/members/working-groups

DATA AVAILABILITY

No datasets were generated or analyzed for this study.

AUTHOR CONTRIBUTIONS

All authors conceived the manuscript as part of the activities of the PICES FUTURE Scientific Steering Committee. SB wrote the article with contributions from EDL, RP, RR, MM, HS, and TT. All authors provided the edits and comments to the manuscript. EDL developed the FUTURE and case study schematics.

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APPENDIX 1: FUTURE SCIENCE RESEARCH THEMES

(1) What determines an ecosystem's intrinsic resilience and vulnerability to natural and anthropogenic forcing?

- (1.1) What are the important physical, chemical and biological processes that underlie the structure and function of ecosystems?
- (1.2) How might changing physical, chemical and biological processes cause alterations to ecosystem structure and function?
- (1.3) How do changes in ecosystem structure1 affect the relationships between ecosystem components?
- (1.4) How might changes in ecosystem structure and function affect an ecosystem's resilience or vulnerability to natural and anthropogenic forcing?
- (1.5) What thresholds, buffers and amplifiers are associated with maintaining ecosystem resilience?
- (1.6) What do the answers to the above sub-questions imply about the ability to predict future states of ecosystems and how they might respond to natural and anthropogenic forcing?

(2) How do ecosystems respond to natural and anthropogenic forcing, and how might they change in the future?

- (2.1) How has the important physical, chemical and biological processes changed, how are they changing, and how might they change as a result of climate change and human activities?
- (2.2) What factors might be mediating changes in the physical, chemical and biological processes?
- (2.3) How does physical forcing, including climate variability and climate change, affect the processes underlying ecosystem structure and function?
- (2.4) How do human uses of marine resources affect the processes underlying ecosystem structure and function?
- (2.5) How are human uses of marine resources affected by changes in ecosystem structure and function?
- (2.6) How can understanding of these ecosystem processes and relationships, as addressed in the preceding sub-questions, be used to forecast ecosystem response?
- (2.7) What are the consequences of projected climate changes for the ecosystems and their goods and services?

(3) How do human activities affect coastal ecosystems and how are societies affected by changes in these ecosystems?

- (3.1) What are the dominant anthropogenic pressures in coastal marine ecosystems and how are they changing?
- (3.2) How are these anthropogenic pressures and climate forcings, including sea level rise, affecting nearshore and coastal ecosystems and their interactions with offshore and terrestrial systems?
- (3.3) How do multiple anthropogenic stressors interact to alter the structure and function of the systems, and what are the cumulative effects?
- (3.4) What will be the consequences of projected coastal ecosystem changes and what is the predictability and uncertainty of forecasted changes?
- (3.5) How can we effectively use our understanding of coastal ecosystem processes and mechanisms to identify the nature and causes of ecosystem changes and to develop strategies for sustainable use?

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ORIGINAL RESEARCH
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Redmap Australia: Challenges and Successes With a Large-Scale Citizen Science-Based Approach to Ecological Monitoring and Community Engagement on Climate Change

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Citizen science includes a suite of research approaches that involves participation by citizens, who are not usually trained scientists, in scientific projects. Citizen science projects have the capacity to record observations of species with high precision and accuracy, offering the potential for collection of biological data to support a diversity of research investigations. Moreover, via the involvement of project participants, these projects have the potential to engage the public on scientific issues and to possibly contribute to changes in community knowledge, attitudes and behaviors. However, there are considerable challenges in ensuring that large-scale collection and verification of species data by the untrained public is a robust and useful long-term endeavor, and that project participants are indeed engaged and acquiring knowledge. Here, we describe approaches taken to overcome challenges in creation and maintenance of a website-based national citizen science initiative where fishers, divers, and other coastal users submit opportunistic photographic observations of 'out-of-range' species. The Range Extension Database and Mapping Project (Redmap Australia) has two objectives, (1) ecological monitoring for the early detection of species that may be extending their geographic distribution due to environmental change, and (2) engaging the public on the ecological impacts of climate change, using the public's own data. Semi-automated 'managed crowd-sourcing' of an Australia-wide network of scientists with taxonomic expertise is used to verify every photographic observation. This unique system is supported by efficient workflows that ensures the rigor of data submitted. Moreover, ease of involvement for participants and prompt personal feedback has contributed to generating and maintaining ongoing interest. The design of Redmap Australia allows co-creation of knowledge with the community – without participants requiring formal training – providing an opportunity to engage sectors of the community that may not necessarily be willing to undergo training or otherwise be formally involved or engaged in citizen science. Given that capturing changes in our natural environment requires many observations spread over time and space, identifying factors and processes that support large-scale citizen science monitoring projects is increasingly critical.

Keywords: citizen science program, climate change ecology, community-based monitoring, data verification, range-shift, science communication, species identification, species redistribution

INTRODUCTION

Interest in natural history citizen science projects has proliferated in recent years, with the majority of projects aiming to explore a wide variety of biological and ecological issues (Silvertown, 2009). Citizen science data have been particularly useful for documenting poleward changes in distribution ('range shifts') for numerous taxa across the world, providing some of the strongest evidence that species are responding to recent climate change (Parmesan and Yohe, 2003). Although citizen science is not a new concept, research involving public participation has also flourished in recent years (Silvertown, 2009; Theobald et al., 2015), including in marine systems (Thiel et al., 2014). Citizen science projects aimed at ecological monitoring can be classified along a spectrum (Shirk et al., 2012) from programs that have a small number of highly trained contributors (e.g., Reef Life Survey[1]), where the data obtained can be as accurate as that from professional scientifically trained observers (Edgar and Stuart-Smith, 2009), through to those with a comparatively larger number of contributors, but with (generally) little or no training (e.g., Project Noah[2]). Projects operating at a larger scale and without extensive formal training of contributors are often criticized about the scientific credibility of observations submitted. Use of these datasets by scientists and resource managers may be hindered by a perception they are of lesser quality compared to those collected by scientists (Bird et al., 2014). The challenge is to create and maintain a successful citizen science project that facilitates long-term, large-scale, and robust data collection (Bonney et al., 2009), ensures motivated and satisfied project participants (Wald et al., 2016), and creatively engages the broader public with dialogue on potentially confronting issues (Martin, 2017), like climate change.

In common with many regions around the world, monitoring for changes in the distribution of marine species along Australia's 60,000 km of coastline is an ongoing challenge, particularly given significant funding constraints (Tulloch et al., 2013), and the often limited baseline data available (Gledhill et al., 2015). The urgency of collecting data on climate-driven shifts in species distributions may be diminished given variable public acceptance of climate change (Leviston et al., 2013), and poor acknowledgment of climate impacts within some marine industries (Nursey-Bray et al., 2012). This is despite Australian marine waters warming at a rate of at least two times the global average along the northern half of the country, three times the global average in the south–west and nearly four times in the south–east (Hobday and Pecl, 2014). This has been associated with extensive biological changes documented throughout marine ecosystems that are consistent with those expected under climate change (e.g., Poloczanska et al., 2007; Pecl et al., 2017, 2019). However, Australia has several million marine resource users, for example indigenous and commercial fishers, and recreational users such as divers, fishers, and beachcombers (Gledhill et al., 2015), that regularly participate in coastal activities and could provide observations of their environment.

Redmap Australia (the Range Extension Database and Mapping Project[3]) is a citizen science Project inviting members of the Australian public to submit photographs and associated data about unusual observations of marine species made while undertaking activities like fishing, diving, boating, and beachcombing. The project started in the Island state of Tasmania in 2009, before expanding nationally across Australia at the end of 2012. Redmap has two linked and equally important aims, (1) ecological monitoring to provide an early indication of potential range shifts, and (2) actively engaging the broader community on issues regarding marine climate change, largely using their own data. At a global scale, quantitative assessments of 'out-of-range' marine species observations show that these are strongly associated with longer-term climatic features (García Molinos et al., 2015), i.e., they are not merely random observations of 'vagrant' individuals (Fogarty et al., 2017). Repeated or consistent out-of-range observations of individuals, in regions not previously observed, therefore likely represent the 'arrival' stage of range-shifting species into a new or extended area of their range (Bates et al., 2014). Thus, out-of-range observations

[1]www.reeflifesurvey.com
[2]www.projectnoah.org
[3]www.redmap.org.au

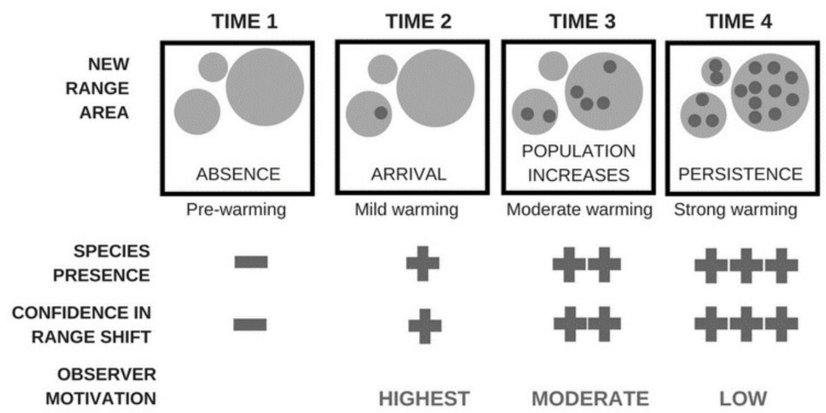

FIGURE 1 | The stages of a species geographic range extension, with gray circles representing possible suitable habitats and black circles individuals of a given species (adapted from Bates et al., 2014). Out-of-range observations are associated with longer-term climatic features (Fogarty et al., 2017), i.e., they likely represent the 'arrival' stage of a range extension and it is these individuals that the Redmap project is designed to detect. When species presence is low at a given location, citizen scientists are motivated to report species occurrences, however, this motivation declines after it becomes more widely known and accepted that a species now occurs in a particular area. Confidence that a range shift has occurred increases over time, as individuals of a species persist and continue to be recorded within a given region.

of species may provide an early indication of impending range shifts (aim 1 above and see **Figure 1**).

Here, we share some of the key challenges faced, and corresponding solutions implemented, in establishing, operating, and maintaining a national scale citizen science project with the objective of documenting observations of potential range-shifting marine species along the Australian coastline (**Figure 2**). Our aim is to share with other practitioners of citizen science our experiences and lessons learned. We articulate how we developed a project that was (1) feasible over large-scales involving multiple institutes, and could potentially operate at relatively small cost and time investment, yet still (2) achieved robust observational data that could be used with confidence by scientists and resource managers. The system also needed to be (3) flexible so the project can adapt over time. Lastly, we needed (4) creative methods for engaging with a large and geographically dispersed community of potential citizen scientists. This description of our approach to achieving the collection of credible species data, based on relatively rare observations and from a (potentially) untrained public, and in successfully engaging the public on a confronting issue like climate change, may be useful in other citizen science or research contexts.

MATERIALS AND METHODS

Redmap's citizen scientists can use region-specific lists of 'target' or 'listed' species (i.e., species that may be extending their distribution into 'new' areas) available on the website, or smartphone application (available at Google Play and iTunes), to help identify which species may be unusual to their particular area before logging an observation. However, contributors can also submit photographs of any species they consider unusual for their area (referred to as 'other' species). Photographs of observations are sent to members of a panel of 80 expert scientists from 26 institutions across the country to verify species identification. After verification, observations are displayed on the website[4] and the contributor is sent individual feedback via a system-generated, but personalisable and editable, email (sent by the verifying scientist through the verification system) (see section "Ensuring Robust 'Out-of-Range' Observational Data" and **Supplementary Table S1**).

The Redmap model was designed to provide a framework for efficiently collecting, collating, verifying, sharing and using geo-referenced species observational data. It was critical that 'opportunistic' participation in the project was easy by (potentially) untrained members of the public, whilst rigorous, standardized, and transparent *post hoc* verification of submitted observations was still achieved. Australian marine users have expressed a strong preference to participate in data collection more than any other role in marine research (Martin et al., 2016a), designing projects, establishing project goals or interpreting results for example, making a 'contributory' citizen science project (Roy et al., 2012) a good match to the public's interest in marine data collection.

Implementing a Large-Scale Low-Cost Collaborative Project

The collection and processing of species observational data at the national scale presented several challenges including:

(a) Governance of the project in a multi-jurisdictional environment.
(b) Timely management of observations submitted by the public.
(c) Maximizing efficiency and minimizing operational costs under financially constrained circumstances.

[4] http://www.redmap.org.au/sightings/

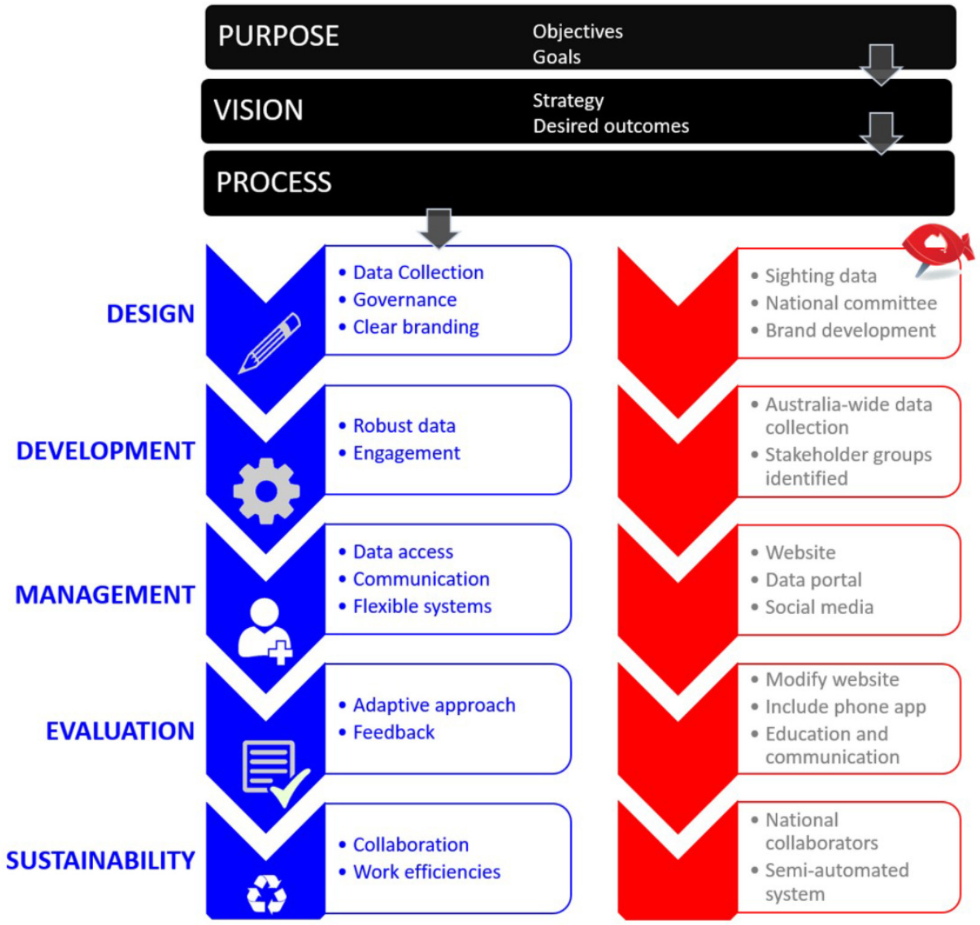

FIGURE 2 | The objectives and goals of Redmap have determined the project strategy and the desired outcomes, which in turn have defined the approaches taken (right in red) to address the key challenges associated with development and maintenance of Redmap as a successful large-scale long-term citizen science project (left in blue).

(d) Identifying target species and managing information for species from across a large geographic area.

Operation of the project at a national scale required the cooperation of regional organizations who could work locally to lead and promote the project and provide local resources for species selection (see section "Ensuring Robust 'Out-of-Range' Observational Data") and identification. Governance is based on a distributed ownership but centrally managed model. A lead organization in each state is responsible for promoting the project and is assigned ownership of all regional data. A governance model was agreed upon among all participating organizations that included:

(a) Establishing a National Steering Committee to oversee and direct the project.
(b) Developing a memorandum of understanding among the multiple organizations, in particular to deal with governance of data.
(c) Establishing committees to support the two main objectives of Redmap – the Science and Data Advisory Panel and the Community Engagement Advisory Panel.
(d) Appointing the role of National Co-ordinator, supported by Regional Administrators in each jurisdiction.

Ensuring Robust 'Out-of-Range' Observational Data

Region administrators worked with marine biogeography experts and community members to develop a list of target species (i.e., species that may be extending their distribution into 'new' areas), highlighted on the website and smartphone application. The initial species lists were collated by a range of marine users in each region including scientists, resource managers, fishers, and divers, who all agreed on proposed species as ones they had not seen previously in each respective area. We then used this information in combination with peer-reviewed literature, species observation databases and museum-based checks on species distributions to have some level of certainty of what the known (historical) distribution was. Redmap species ranges were then formally defined using multiple reference sources for each species (at least three for each species) using published literature and online resources (see Stuart-Smith, 2017). In addition to having a well-defined distribution, species selection was also contingent on a number of factors including the ability to easily identify the species from an image only (i.e., with no easily confused similar species) and being easily observable (i.e., not cryptic or camouflaged).

A distributed automatic system was designed to workflow the sightings to the appropriate administrator and allow 'managed crowdsourcing' of scientists for data verification and processing (see **Figure 3**). The system can define any number of regions (defined by state or territory) and an administrator is assigned to manage each region. For data verification, scientists are assigned to each species in the database, according to taxonomic or ecological expertise. A single species can have multiple scientists assigned, allowing for situations when a scientist may not be available or does not respond in a timely manner. Additionally, the location of the observation provided by the user is automatically compared to the known and pre-defined distribution for a species and scientists may nominate to only receive observations that are determined to be 'out of range.' If scientists nominate to only receive out-of-range observations, in most regions, in-range observations are checked by an administrator, and then displayed on the web site. The system also allows for submission of observations of non-listed species deemed by a member of the public to be potentially "unusual"; these are not automatically assigned to scientists for verification and are instead assigned to the Regional Administrator before being reassigned to scientists with appropriate taxonomic expertise.

Contributors submit photographs through web forms or smartphone applications and an automated workflow sends an email to the assigned scientist. The email contains a link to the Redmap website where the scientist can verify the observation based on a predetermined set of conditions (**Supplementary Table S1**), ensuring a uniform method is used to verify observations. This system increases efficiency by minimizing data handling time for individual observations and distributing the workload across multiple scientists. The observation data flows through a series of mostly automated steps (**Figure 3**) whereby condition matrices or manual input by scientists or regional administrators dictate the outcome of an observation.

Flexible Systems for an Adaptive Approach

While smaller and shorter term (<3 years) citizen science projects are often successful due to the full–time efforts of a small group of individuals, projects that have longer (>5 years) and larger scale ambitions need to be adaptive in relation to operational and structural aspects. The Redmap online platform was designed for flexibility and easy upscaling or downscaling where required, enabling more species or verification scientists as needed to be added to the system (see **Figure 3**). Presently 220 species are listed and approximately 80 verifying scientists, however, there is no limit to the number of species or scientists that could be accommodated by the platform. The system can also accept additional regions, countries, or habitats (e.g., freshwater or terrestrial). A centralized species database allows multiple regions to have the same species identified on their regional site for logging without replicating content.

Engaging Effectively With a Large and Geographically Dispersed Community

Since its inception as a pilot project in the state of Tasmania in late 2009 (and expanding nationally in late 2012), Redmap has placed equal weight on the dual goals of ecological monitoring for early detection of potentially range shifting species and engagement and communication with the community about marine climate change. Communication strategies were designed to incorporate elements of dialogue and participation to ensure effective and best practice science engagement (Groffman et al., 2010). Dialogue occurs through interaction of the citizen scientist with the verifying scientist via email, regular interactions with the Redmap project via the website and social media (Facebook page, Twitter and Instagram), and through participation in regional events and presentations to public and community groups. The tone of the Redmap message plays an important role in public image, and strives for a balance between providing scientific facts, asking for assistance, and acknowledging the value added by contributors. Additionally, Redmap team members regularly participate in fishing competitions and dive club trips, and give presentations at club meetings or public events. These activities not only allow targeted promotion of the project but also facilitate close ties between the scientists and the broader community enabling one-on-one discussion.

Respecting Citizen Scientists as Data Owners

An essential element of securing ongoing community engagement in an online program such as Redmap, is being clear on whether and how data will be made publicly available, and to what extent it will comply with the principle of 'freely available' (Zapico, 2013). Although many ocean users are forthcoming and prepared to share information, some may resist providing location details of their favorite fishing or dive spots (Bannon, 2016). Such concerns impact the willingness of people to participate and contribute data; Redmap respects contributors' need for confidentiality and gives participants control and discretion to select a range of spatial scales to record their observation. Additionally, although data are displayed on the website immediately after verification, Redmap data are embargoed from direct bulk download providing exact reported latitude and longitude for 3 years. Similarly, the quality and accessibility of underwater photography equipment has aided the proliferation of amateur and semi-professional underwater photographers, and some photographers may be concerned about loss of ownership of their photograph. When submitting an observation, the contributors' permission is sought to use the photograph on the web site and only for promotional not-for-profit purposes.

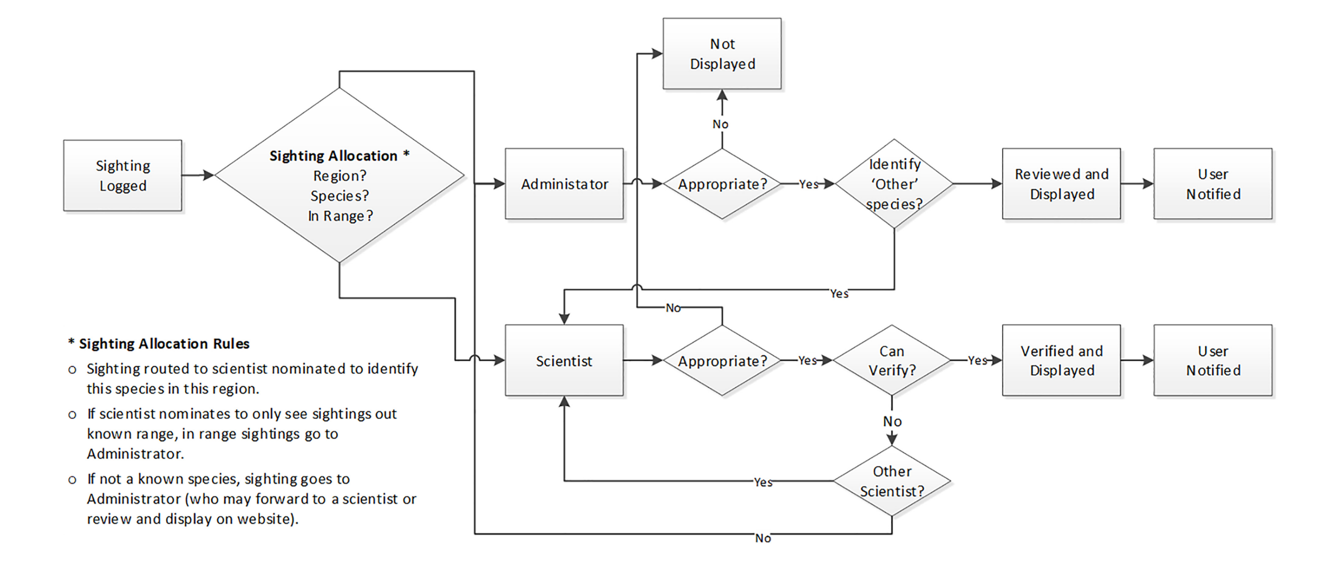

FIGURE 3 | Overview of the management workflow for observations submitted for the Redmap website and database verification system, including feedback provided to the contributor, and the flexible 'sighting allocation' process that is key to the semi-automated observation verification system.

Feedback to Contributors and the Wider Community

Redmap provides instantaneous feedback to everyone who logs an observation, in the form of a thank-you email which informs them about the next step in the verification process (**Figure 3**). Group-level feedback to contributors and the broader community is provided through messages celebrating the latest observations, or the group's achievements (e.g., total observations, or other records), via social and online media, as well through a digital newsletter and at public presentations. Redmap's website was designed specifically to provide information in a dynamic way with minimal input by the project administrators. The home page is a continuously updated platform which automatically displays the most recent observations. Verified observations are loaded onto an interactive map of Australia, so site visitors can see who has logged the most recent observations and where.

RESULTS

Robust Observational Data to Provide an Indication of Potential Range Shifts

In a relatively short time for an ecological monitoring project, Redmap has delivered on the project's scientific and engagement objectives. To date, 2,078 observations have been submitted by 784 citizen scientists and verified by 59 scientists (**Figure 4**). Given the project is very specifically and only interested in 'out-of-range' or unusual observations, and relies on participants identifying that a given species occurrence is unusual, this is a significant and unique data contribution. Redmap is after 'outliers,' not all or any species observed, such as in a survey. It is only the 'arrival' stage of potential range-shifters that the project is interested in (**Figure 1**), as an early indication of which species are potentially shifting distribution (Fogarty et al., 2017), or responding to anomalous events like marine heatwaves.

Among the scientific deliverables, the Redmap team developed a qualitative rapid assessment tool to classify levels of confidence (i.e., high, medium and low) in records of potential range extension for Redmap-listed species in Tasmania (Robinson et al., 2015). The assessment built on methods used in the early detection of invasive species using two systematic and transparent qualitative decision trees to determine both the confidence with which we could define the historical range boundary for each species and the strength of evidence for a species being consistently detected out-of-range, considering mobility, detectability and temporal and seasonal consistency of observations. Combining the outputs of both decision trees allowed designation of the 'confidence' in a potential range extension of each species, providing evidence of poleward range extensions beyond previously reported range boundaries for multiple species of fish and invertebrates (Robinson et al., 2015). In consultation with many of the fishers and divers that submitted observations to Redmap, this assessment was drafted and published as a 'report card' for public community dissemination of the project results[5] and as a journal article for the scientific community (Robinson et al., 2015). Similar assessments are now being conducted for other parts of the eastern and western Australian coastline.

Redmap has made small, but influential, data contributions to a number of studies (Johnson et al., 2011; Last et al., 2011; Madin et al., 2012; Grove and Finn, 2014; Ramos et al., 2014, 2015; Couturier et al., 2015; Stuart-Smith et al., 2016). These works, together with the Robinson et al. (2015) study, have been instrumental in developing a strong understanding of species and ecosystem changes occurring in south–east Australia, one

[5] www.redmap.org.au/article/the-redmap-tasmania-report-card/

of the regions which is warming in the top 10% for rate of warming globally (Hobday and Pecl, 2014). Frequently reported 'out-of-range' species in Tasmania include zebrafish (*Girella zebra*), yellowtail kingfish (*Seriola lalandi*), white-ear (*Parma microlepis*), herring cale (*Olisthops cyanomelas*), luderick (*Girella tricuspidata*), old wife (*Enoplosus armatus*), and snapper (*Pagrus auratus*), many of which have been recorded over multiple years, in cooler winter months, and at a variety of life stages. Juveniles of potential range-extending species recorded in colder months are particularly important as they indicate the prospect of species being able to survive throughout the year, increasing their likelihood of reaching adulthood, reproducing, and establishing a population (**Figure 1**; Bates et al., 2014). The eastern rock lobster *Sagmariasus verreauxi* (**Figure 4**) is to date the most logged species on Redmap with "high" confidence of potentially being a range extending species (Robinson et al., 2015). Although this species has been recorded intermittently in Tasmanian waters over several decades, Redmap has now documented large groups (35+ individuals) of adults and sub-adults farther polewards of their accepted range.

Redmap observations have also triggered focused studies on particular species, providing an indication of where limited resources for research could be constructively directed. For example, after out-of-range observations of *Octopus tetricus* were reported to Redmap, the project team made enquiries revealing that this species was now 13% of a local octopus fishery in a suspected 'new' part of its distribution. Subsequent research clearly demonstrated that this species was now present consistently throughout the year (Ramos et al., 2014), was reproductively viable at the new range edge (Ramos et al., 2015) and also provided strong genetic evidence to support a rapid and recent range extension (Ramos Castillejos, 2015). Lastly, many of the sightings submitted to Redmap that are not designated as technically 'out-of-range' have been valuable for improving our knowledge of the distribution of poorly known or rare species, and Redmap has contributed species distribution data to the Australian Faunal Directory[6].

Engaging the Broader Marine Community Using Their Own Data

In addition to effective engagement being one of the projects key goals, it was also essential for reaching and retaining the geographically dispersed and large audience that may potentially recognize and observe the rare occurrence of an out-of-range species. Our target audiences use a variety of information sources regarding the marine environment, like Facebook, other internet sources, government agencies and face-to-face communication (Martin et al., 2016a), and so it is critical that we use a range of avenues for communication. Direct feedback from a scientist is considered the most important element in the Redmap/contributor relationship (Martin et al., 2016a). Our citizen scientists receive immediate email acknowledgment after submitting an observation, followed by the second stage of observer feedback usually within a week of the observation being logged (i.e., species confirmation by verifying scientist, **Figure 5**). Our formal project evaluation suggested that this one-on-one dialogue between scientists and observers helped observers build trust in the project, and makes the participants feel valued and that their contribution is important (Nursey-Bray et al., 2017).

In terms of broader communication, distributed ownership in the project helps ensure wide reach within the community, and this, combined with centralized oversight, achieves a uniform approach in branding and economies of scale where possible. Redmap reaches a large audience – the Redmap Facebook page has >9,400 followers reaching upwards of 50,000 people each month. The website has had >1,000,000 web page downloads with visits from >180 countries, and there has been at least 260 media or outreach articles, and over 165 events or presentations related to the project. The electronic newsletter is sent 2–4 times a year, depending on funding, and reaches 3,000 direct subscribers. The science and communication efforts of the project team have been recognized with a suite of regional, national and international awards and nominations (**Supplementary Table S2**).

Two studies have specifically evaluated the effectiveness of the communication and engagement efforts of the Redmap project (Bannon, 2016; Nursey-Bray et al., 2017). People interacting with Redmap reported that they had learnt about potential range extensions, fish species and distributions, and insights into the relevance of having accurate species information incorporated into policy decisions. Almost half of survey respondents felt they had learnt new information, and 97% of people indicated they trusted the data and information emerging from Redmap (Nursey-Bray et al., 2017). Detailed face-to-face interviews with marine resource users, where Redmap was not mentioned *a priori* to interviewees, strongly suggest that Redmap has been very effective in building awareness and understanding of marine climate change in the community (Bannon, 2016).

DISCUSSION

Citizen science programs have diverse objectives, however, delivering authentic and robust scientific outcomes is a common objective (Tulloch et al., 2013). Moreover, the long-term activity of many citizen science projects requires meeting both scientific and engagement objectives to ensure on-going support from funding bodies and host institutions, as well as continuing participation from the contributing scientists and citizen scientists.

Many biodiversity-based citizen science programs involve recording non-target species over broad geographic regions to address a variety of *a posteriori* research questions (Dickinson et al., 2010). In contrast, Redmap data are collected with *a priori* specific mechanism-driven predictions in mind. In this case the hypothesis-driven objective is the early detection of poleward shifts of species, and the project has been successful in that regard, contributing extensively to our understanding of the changes in the distributions of many species, including fishes, crustaceans, and molluscs, occurring along parts of the Australian coastline. Redmap is analogous to monitoring programs implemented to detect invasive species that have clearly demonstrated the value

[6]http://www.environment.gov.au/science/abrs/online-resources/fauna

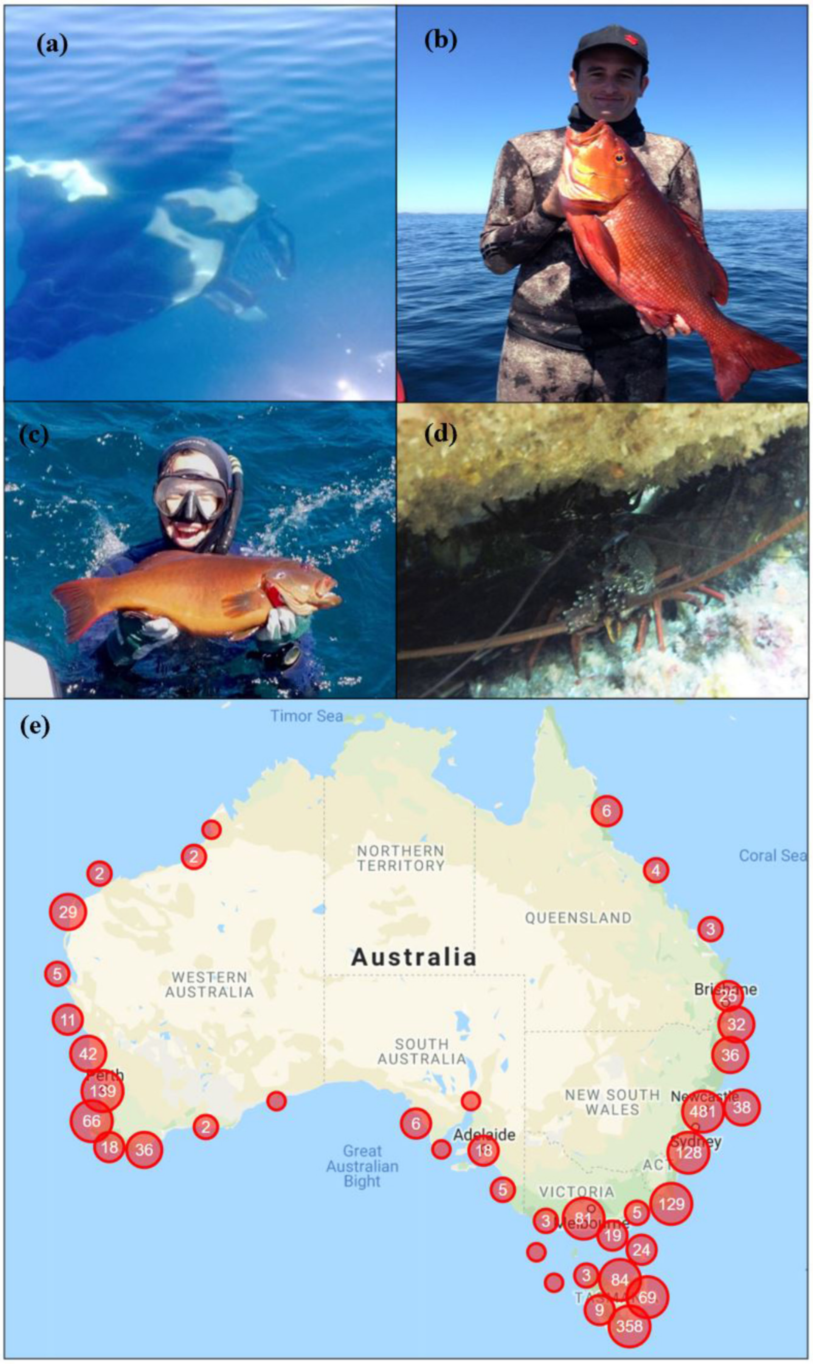

FIGURE 4 | Observations of species reported outside of their known range by Redmap contributors: **(a)** Manta ray, *Manta birostris*, reported in north–east Tasmania (www.redmap.org.au/sightings/1191/) by Leo Miller. This is thought to be Australia's southernmost observation of the species, with the next most southerly observation just south of Sydney (Couturier et al., 2015, Photograph: Leo Miller). **(b)** Red bass, *Lutjanus bohar*, caught off Newcastle NSW (www.redmap.org.au/sightings/1628/). In Australia this species in known to occur in Western Australia, and on the east coast from the Great Barrier Reef to southern Queensland (Photograph: Kylie Johnston). **(c)** Coral trout, *Plectropomus leopardus*, caught off Sydney (www.redmap.org.au/sightings/1624/). In Australia its distribution occurs from north-western Western Australia, around the north of Australia to the southern end of the Great Barrier Reef, in Queensland on the east coast (Photograph: Derrick Oscar Cruz). **(d)** A den of eastern rock lobster, *Sagmariasus verreauxi*, photographed at St Helens off eastern Tasmania (www.redmap.org.au/region/tas/sightings/1238/). This species is more common to New South Wales but is observed or captured occasionally along the north coast of Tasmania in small numbers. Larvae are thought to be delivered into Tasmanian waters via the East Australian Current (Photograph: John Keane). **(e)** Locations of observations submitted to Redmap by citizen scientists over the duration of the program. Numbers within the red circles indicate the number of discrete observations submitted for that area. People depicted within the images gave their written informed consent for the photographs to be included.

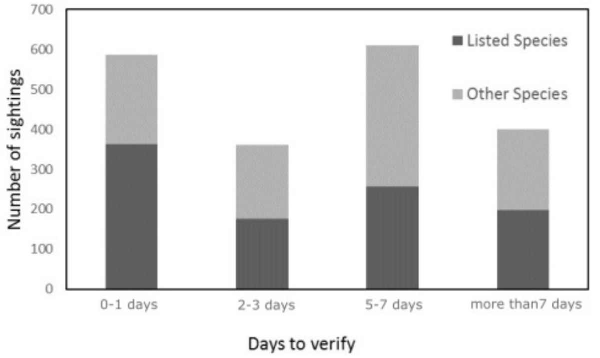

FIGURE 5 | Number of days taken to verify species identification after an observation is submitted to Redmap Australia via the website or smartphone app. Note, 'other species' are species not 'listed' as Redmap species, and therefore do not have existing database links directly to expert scientists, instead being directed to experts manually via the regional administrator.

of engaging citizens for assessing and responding to large-scale and time-sensitive conservation problems (Scyphers et al., 2014). However, although citizen science programs with greater spatial coverage are useful for biodiversity research (Tulloch et al., 2013; Theobald et al., 2015), there are significant challenges in developing and maintaining such programs over both large temporal and spatial scales.

One of the most influential factors allowing implementation of the project has been increased access and use of wireless technologies in the community (Silvertown, 2009). Within the Redmap project, integration of the website, smartphone application, and social media (Facebook, Instagram, and Twitter) have been instrumental in maximizing reach, streamlining workloads and processes, and allowing collaborators to work together. The website supports collaborative relationships by allowing separate webpages for each region (with news stories, relevant articles, and region-specific observations) all controlled by the lead organization in each region. This capability (and flexibility) allows for different interests among regions to be accommodated within the project as a whole as well as addressing the need to identify key demographic groups and tailor the user experience accordingly. One of the challenges and costs of using websites and apps is the continual changes in the platforms, as developers update software this cascades in updates to websites and apps that require money and time to manage. For projects to maintain a presence over many years requires allocation of funds and planning to maintain a presence.

Despite the significant rewards associated with large-scale collaborations, the potential challenges and costs need to be taken into consideration when forming these projects and the associated networks. Elements that need to be considered are sourcing appropriate people and ensuring that time and funds are made available to establish and maintain the collaboration. This time and effort allows collaborators to discuss and mitigate risks associated with brand dilution, consistency in agreed project objectives, data ownership, legal obligations, and dealing with unforeseen sensitive issues. Major challenges for large-scale citizen science programs like Redmap include needing to secure ongoing funding so that the ecological value of data increases over time, ideally over several decades (Theobald et al., 2015). However, the expectations of funders or host organizations is for clear scientific or community outcomes after 1–3 years of investment, which requires both short and long-term goals to be identified in the project, increasing the engagement of funding bodies and supporters. Use of data in scientific publications or policy action maintains interest and retention of contributors, as volunteers perceive their efforts are leading to something of value (Thiel et al., 2014; Martin et al., 2016b).

Semi-automated workflows for scientific verification of observations are critical mechanisms that allow Redmap to continue functioning when funding reductions preclude project staffing, but the project is also able to implement several other mechanisms to help with this. This includes the ability to archive observations of non-listed 'other' species that require manual processing by a regional administrator (for later processing) and the current development of a "Redmap Champions" package to train volunteer citizen scientists in promotional roles across the country, such as staffing educational booths at public events, drafting articles for online newsletters or presenting talks on the project to various fishing or diving clubs. This package outlines the goals and key messages, project sensitivities, and observation instruction guidelines. Engaging volunteers to act as 'champions' satisfies the interests of many marine users who are keen to communicate citizen science findings (Martin et al., 2016a) and will allow continuation of project promotion and on-ground activities (such as presentations), in the light of potential periods of restricted funding.

The educational resources developed and provided by the Redmap project on the website are a valuable part of engaging the whole community about the environment and the impacts of climate change. These resources generate a sense of community ownership of the project with Redmap community members producing articles, delivering presentations on the project and its outputs, and assisting greatly with dissemination of Redmap project materials. Whilst Redmap is primarily a 'contributory' citizen science project (Roy et al., 2012), it includes elements of a 'collaborative' model where members of the public not only contribute data, but also inform the way in which the questions are addressed and disseminate findings (Roy et al., 2012). For example, members of the fishing and diving communities were actively involved in the selection of species being monitored, and in the design, testing and production of various aspects of the website, smartphone application and major communication outputs like the 'Redmap Tasmania Report Card'[7].

For Redmap, community knowledge is embedded in the contributors through their observations of "something different" or more specifically the observation of something uncommon or new. For scientists, this knowledge fits into a system associated with a changing climate and the possible polewards range extension of species. The knowledge exchange provides the basis for observation and explanation, answering people's questions about particular species whilst highlighting to the users that their

[7] www.redmap.org.au/article/the-redmap-tasmania-report-card/

observation is useful and they are making a valued contribution to science. In this way, Redmap presents participants with a valuable opportunity to interact with scientists directly – an important design consideration for improving the relationship between science and society (National Academies of Sciences, Engineering, and Medicine., 2017).

Only half of marine citizen science projects include quality control to ensure the collection of a robust dataset that can be defended (Thiel et al., 2014), despite this being critically important to the credibility of the data (Ratnieks et al., 2016). Consequently, a challenge to citizen science projects in general is the acceptance of volunteer-collected data by the scientific and resource management communities (Dickinson et al., 2010) Building on early and significant successes from bird-based citizen science endeavors (Silvertown, 2009), the recent surge in high quality research papers using citizen-science data (e.g., Stuart-Smith et al., 2013) and development of statistical options available for handling common biases (e.g., Bird et al., 2014) have paved the way to making volunteer data more accepted by the scientific community.

Key to Redmap's success has been the rapid verification and archiving of every photographic observation submitted to Redmap only made tractable by the 'managed crowd-sourcing' of taxonomic experts across the country. This semi-automated process significantly reduces contributor error in species identification, and importantly allows submission of data from (potentially) untrained participants. Moreover, the easy and opportunistic involvement of participants, irrespective of training or knowledge, in combination with our data verification processes, highly efficient workflows, and quality detailed personal feedback to contributors, has laid the foundation for a large scale monitoring project that has already produced tangible outputs and established a legacy that will maximize the longevity of a project that will be increasingly critical as changes in the marine environment due to climate change escalate.

AUTHOR CONTRIBUTIONS

GP conceived the idea for Redmap Australia and designed the methodology primarily with JS-S, PW, OG, GJ, and NM with contributions from a broader group. GP led writing of the manuscript which included contributions from all authors. All authors contributed critically to the drafts and gave final approval for publication.

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Coral Translocation as a Method to Restore Impacted Deep-Sea Coral Communities

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Corals and sponges in rocky deep-sea environments are foundation species postulated to enhance local diversity by increasing biogenic habitat heterogeneity and enriching local carbon cycling. These key groups are highly vulnerable to disturbances (e.g., trawling, mining, and pollution) and are threatened by expansive changes in ocean conditions linked to climate change (acidification, warming, and deoxygenation). Once damaged by trawling or other disturbances, recolonization and regrowth may require centuries or longer, highlighting the need for stewardship of these deep-sea coral and sponge communities (DSCSCs). To this end, the sustainability of DSCSCs may be enhanced not only by protecting existing communities, but also repopulating disturbed areas using active restoration methods. Here, we report one of the first studies to explore methods to restore deep-sea coral populations by translocating coral fragments of multiple coral species. Branches of deep-sea corals were collected by ROV from 800 to 1300 m depth off central California and propagated into multiple fragments once at the surface. These fragments were then attached to "coral pots" using two different methods and placed in the same habitat to assess their survivorship (n = 113 total fragments, n = 7 taxa, n = 7 deployment groups). Mean survivorship for all translocated coral fragments observed within the first 365 days was ~52%, with the highest mortality occurring in the first 3 months. In addition to an initial temporal sensitivity, survival of coral fragments varied by attachment method and among species. All coral fragments attached to coral pots using zip ties died, while those attached by cement resulted in differential survivorship over time. The latter method resulted in 80–100% fragment survivorship after 1 year for *Corallium* sp., *Lillipathes* sp., and *Swiftia kofoidi,* 12–50% for the bamboo corals *Keratoisis* sp. and *Isidella tentaculum*, and 0–50% for the bubblegum corals *Paragorgia arborea* and *Sibogagorgia cauliflora*. These initial results indicate differences in sensitivities to transplanting methods among coral species, but also suggest that repopulation efforts may accelerate the recovery of disturbed DSCSCs.

Keywords: deep-sea coral restoration, *Corallium* sp., *Lillipathes* sp., *Swiftia kofoidi*, *Keratoisis* sp., *Isidella tentaculum*, *Paragorgia arborea*, *Sibogagorgia cauliflora*

INTRODUCTION

Human impacts in the deep-sea are increasing from direct extractive activities (e.g., fishing, mineral extraction), pollution (e.g., oil spills, trash), and expansive changes in ocean conditions linked to anthropogenic climate change (ocean acidification, warming, and deoxygenation). While the integrated impacts of human activities in the deep-sea remain largely unknown, there is clear evidence that bottom trawling for fishes and invertebrates alone is leaving a global footprint from the nearshore to $>$1000 m depth (Amoroso et al., 2018). Catastrophes such as the 2010 BP Deepwater Horizon oil spill in the Gulf of Mexico also demonstrated that an oil spill from a single platform can have far ranging effects spanning from ecological impacts in the deep-sea to economic depression for the adjacent coastal communities—with the full potential impacts still unknown (McCrea-Strub et al., 2011; White et al., 2011; Sumaila et al., 2012). Although the Magnuson-Stevens Fishery Conservation and Management Act (2006) establishes a legal mandate to minimize negative impacts to these and other ecosystems within the United States Exclusive Economic Zone, the slow rates of recovery for some damaged deep-sea assemblages suggest that active restoration efforts may be beneficial.

Recovery and resilience of impacted deep-sea ecosystems may be enhanced by efforts to propagate species that form key biogenic structures. Species that form biogenic structures are hypothesized to facilitate positive species interactions, further promoting both biodiversity and ecosystem function (Bruno and Bertness, 2000; Bruno et al., 2003). In deep-sea ridge and seamount systems, suspension feeders such as corals and sponges typically dominate the benthos and provide critical biogenic habitat for a variety of associated deep-sea fishes and invertebrates (Rogers, 1999; Koslow et al., 2000; Roark et al., 2005; Stone, 2006; Love et al., 2007; Baillon et al., 2012; Bourque and Demopoulos, 2018). Furthermore, suspension feeding by these organisms may enhance carbon sequestration (Murray et al., 1994; Cathalot et al., 2015; Kahn et al., 2015). Thus, efforts to translocate healthy or rehabilitated corals and sponges may accelerate the recovery of local diversity and ecosystem function in deep-sea coral and sponge communities (DSCSCs) that have been disturbed or destroyed by human activity.

Understanding how to actively enhance or facilitate the recovery of disturbed DSCSCs is a new frontier for restoration science. Van Dover et al. (2014) evaluated hypothetical scenarios of ecological restoration of the deep sea but lacked empirical data from active restoration studies, bringing to light a major gap in applied scientific knowledge. Although experimental restoration of the deep-sea corals *Lophelia pertusa* ($\sim$ 500 m in depth) and *Oculina varicosa* ($\sim$60–120 m) has been somewhat successful in the Gulf of Mexico (Koenig et al., 2005; Brooke et al., 2006; Brooke and Young, 2009), studies examining the sensitivity of multiple coral taxa to differential handling and processing remain a gap in knowledge for all deep-sea efforts. However, insights gained from these initial efforts, in combination with information from coral reef restoration studies from shallow habitats ($<$30 m), may serve as a more comprehensive guide to expanding mitigation strategies for the deep sea.

While studies of both asexual and sexual propagation of shallow water corals are prevalent, we focused on translocating coral fragments from multiple coral taxa as a first step to understanding the feasibility and facilitation of DSCSC recovery in the Eastern Pacific for several reasons. Enhancing coral reef recovery via translocation or transplanting coral fragments are presently proposed to be more advantageous than using sexual propagation methods due to the cost-effectiveness and requirements for technical knowledge (Jaap, 2000; Bowden-Kerby, 2001; Epstein et al., 2001; Spieler et al., 2001; Rinkevich, 2008; Edwards et al., 2010; Villanueva et al., 2012; Barton et al., 2015). That is, propagation of corals by harnessing gametes or larvae requires biological, culturing, and processing knowledge that may be more complicated than fragmenting corals and attaching fragments to a surface. For example, harnessing the sexual propagation of *Acropora* corals in shallow water systems required knowledge of when these corals were likely to broadcast gametes, culturing the gametes through fertilization and larval development, creating larval settlement surfaces, determining the optimal time for larval batch exposure to settlement cues, determining the optimal time for newly settled larvae outplanting, how to outplant or attach the new recruits to a natural reef, and many other steps in the process that required additional resources (Boch and Morse, 2012). Horoszowski-Fridman et al. (2011) investigated the reproductive output of transplanted coral fragments versus natural coral colonies and concluded that the reproductive role of coral transplants needs further attention. Various attachment strategies for scleractinia coral fragments have been reviewed and discussed (Barton et al., 2015) and insights from *in situ* experimental studies with shallow water gorgonians (20–25 m) are also available (Lasker, 1990; Linares et al., 2008a,b). Overall, the survivorship of both fragmented and sexually propagated corals has been shown to be most sensitive during the first post-transplant year and more successful for fragment transplants than sexually propagated corals (Epstein et al., 2001; Boch and Morse, 2012). In over three decades of shallow water coral restoration research, survivorship of asexually propagated corals has typically ranged 30–40% after the first year *in situ*.

In 2013, deep-sea coral and sponge communities (DSCSCs) were discovered at Sur Ridge (36°N; 122°W, **Figure 1A**) by collaborators from the Monterey Bay National Marine Sanctuary (MBNMS, National Oceanographic Atmospheric Administration) and the Monterey Bay Aquarium Research Institute (MBARI). Recognizing the need to establish active deep-sea coral restoration strategies in addition to protecting these highly diverse habitats, the collaborators began coral translocation studies to examine the feasibility of active deep-sea mitigation as an additional option along with protection. Here, we report insights gained from $\sim$3 years of deep-sea coral translocation studies and discuss some of the steps needed to potentially overcome current limitations. As there are no previous publications that report details of active deep-sea coral restoration approaches in the Eastern Pacific, we conducted our translocation experiments to determine the factors that may enhance survivorship by specifically examining coral processing protocols, the sensitivity of various deep-sea coral species to

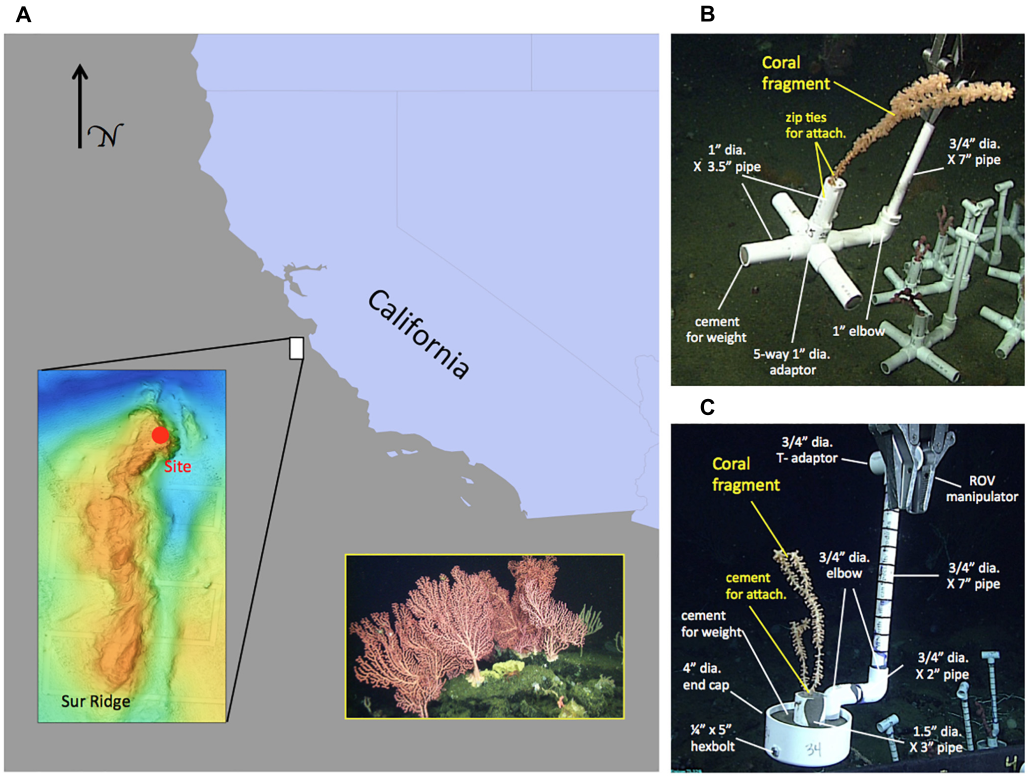

FIGURE 1 | Map of the study area and schematic of coral pots with different coral attachment methods. **(A)** Location of Sur Ridge with respect to California, United States coastline. Lower-left inset: multi-beam sonar map of Sur Ridge. Solid red circle in the sonar map indicates the general location of the study site. Lower-right inset: photograph of an example coral and sponge cluster in the study area taken from the ROV *Doc Ricketts*. **(B)** Example coral pot using the zip tie method for attaching coral fragments in a moveable module. These coral pots (v.1) were also used with the cement attachment method (sans zip ties) in follow-up deployments. **(C)** Example coral pot v.2 using the cement attachment method for attaching a coral fragment in a moveable module. Note: all coral pot materials are Schedule 40 standard poly vinyl chloride (PVC) with the exception of the cement and galvanized steel hexbolt and nut.

translocation stress, and temporal sensitivity in survivorship as an initial foundation for further investigation.

METHODS

Study Site, Environmental Exposure, and General Coral Processing

Coral translocation studies were conducted in DSCSCs at Sur Ridge, a prominent rocky ridge ~60 km west of Point Sur, CA (**Figure 1A**). Coral taxa of interest were identified by the onboard taxonomic experts and using the *Sur Ridge Field Guide* created by the MBNMS and MBARI collaboration[1]. Seven coral taxa (*Corallium* sp., *Lillipathes* sp., *Swiftia kofoidi*, *Keratoisis* sp., *Isidella tentaculum*, *Paragorgia arborea,* and *Sibogagorgia cauliflora;* **Table 1**) common at Sur Ridge were included in the translocation studies.

Collection and translocation studies of coral fragments were conducted using the remotely operated vehicle (ROV) *Doc Ricketts*, operated from the R/V *Western Flyer*—a 36-meter long, small water-plane area twin hull vessel capable of deploying, operating, and recovering tethered remotely operated vehicles for oceanographic research. The *Doc Ricketts* has high-definition cameras, a suite of environmental sensors, robotic arms with manipulators, and storage containers ("bioboxes") for samples. A high-resolution navigation system integrated with the R/V *Western Flyer* enabled repeated visits to specific locations on Sur Ridge for these studies. The data from a SBE 19 CTDO instrument (Sea-Bird Scientific, Bellevue, WA) mounted on the ROV *Doc Ricketts* during all of the ROV dives are reported to show the approximate ambient temperature, dissolved oxygen, and salinity levels experienced by the corals during the descent and ascent of ROV operations, during the collection of the donor coral branches and translocation of processed coral fragments. A sample temperature series logged within a biobox is also reported to provide a proxy of temperature experienced by the coral fragments within the biobox relative to the ambient temperature measurements.

Branches of selected coral species were collected from one or more colonies *in situ* that appeared to be visually "healthy" and stored in bioboxes for transport to the surface. After the ROV was

[1]https://sanctuaries.noaa.gov/science/conservation/sur-ridge-field-guide-monterey-bay-national-mairne-sanctuary.html

TABLE 1 | Metadata for deep-sea coral translocation study.

Collection Date	Deployment Date	Taxa	Fragments#	Approx. length (cm)	Group	Attachment version
7/25/14	7/25/14	*Keratoisis* sp.	10	10–20	A	1
7/28/14	7/28/14	*Isidella tentaculum*	7	10–20	B	1
7/28/14	7/28/14	*Paragorgia arborea*	8	10–15	B	1
6/3/16	6/3/16	*Isidella tentaculum*	10	10–20	C	2
6/3/16	6/3/16	*Paragorgia arborea*	5	10–15	C	2
6/3/16	6/3/16	*Sibogagorgia cauliflora*	5	10–20	C	2
6/4/16	6/4/16	*Keratoisis* sp.	9	10–20	D	2
8/27/16	8/28/16	*Isidella tentaculum*	4	10–20	E_a	2
8/27/16	8/28/16	*Paragorgia arborea*	4	10–15	E_a	2
8/27/16	8/28/16	*Sibogagorgia cauliflora*	2	10–20	E_a	2
8/28/16	8/28/16	*Isidella tentaculum*	4	10–20	E_b	2
8/28/16	8/28/16	*Paragorgia arborea*	4	10–15	E_b	2
8/28/16	8/28/16	*Swiftia kofoidi*	2	10–15	E_b	2
8/7/17	8/8/17	*Keratoisis* sp.	4	10–20	F	2
8/7/17	8/8/17	*Lillipathes* sp.	6	10–15	F	2
8/7/17	8/8/17	*Paragorgia arborea*	10	10–15	F	2
8/8/17	8/8/17	*Corallium* sp.	5	10–15	G	2
8/8/17	8/8/17	*Lillipathes* sp.	4	10–15	G	2
8/8/17	8/8/17	*Paragorgia arborea*	5	10–15	G	2
8/8/17	8/8/17	*Swiftia kofoidi*	5	10–15	G	2

From left to right column: donor coral collection dates, coral pot deployment date, type of coral taxa (n = 7 unique taxa total), the number of fragments or coral pots deployed (n = 113 total coral pots), the size range in length of coral fragment used (centimeters), group denotes the deployment designation, and version of attachment used to fix coral fragments in coral pots (1 = zip tie; 2 = cement). Shaded rows indicate corals held overnight in shipboard aquaria.

recovered shipboard, branches of coral colonies (*Corallium* sp., *Lillipathes* sp., *Swiftia kofoidi*, *Keratoisis* sp., *Isidella tentaculum*, *Paragorgia arborea,* and *Sibogagorgia cauliflora*) were cut into smaller fragments using stainless steel scissors and attached to transportable modules ("coral pots") that facilitated ROV operations with the bioboxes. Each coral fragment was placed approximately 1 inch (∼2 cm) in depth at the bases of the fragments. Each coral pot holding a single coral fragment was labeled with a unique number and deployment group letter (**Figures 1B,C**). Once processed, coral pots were returned by ROV to the site of collection within Sur Ridge DSCSCs to avoid confounding the potential effects of translocation with other factors (**Figure 1A**). Overall, n = 8 translocated coral groups were deployed with a mix of coral taxa. Collection, translocation, and re-visitation of translocated deep-sea corals were executed by careful coordination among the ship's crew, ROV pilots, and scientists.

Coral Fragment Attachment

Survival of coral fragments may be affected by how coral fragments are attached to a transportable substrate or module. To evaluate the effects of attachment methods on translocated coral survivorship, coral pots were fabricated using polyvinyl chloride (PVC) plumbing parts with two different attachment methods (∼US$20 per pot). For attachment version 1 (coral pot v.1), fragments from multiple species were attached to a PVC base using four zip ties which "loosely" fixed the fragments to the upright 1″ PVC in the center of the coral pot (**Figure 1B**). PVC cross-legs at the pot base were filled with Sakrete® fast setting cement patcher (Charlotte, NC, United States) to function as the weight—the total pot without coral weighed approximately 1 kg. For attachment version 2 (coral pot v.2), coral fragments from multiple species were "hard" fixed to the upright 1.5″ PVC in the center of coral pot using the fast setting cement patcher (**Figure 1C**). The 4″ diameter PVC end cap at the base was filled with cement to anchor and enclose the pot in an upright orientation. The volume of cement required to affix fragments to the center PVC varied with the size and taxon of fragments. Version 2 pots weighed approximately 1.2 kg in dry weight without coral fragments.

Transport Stress

Survival of translocated corals may also be particularly affected by the duration of exposure to potentially stressful environmental changes during transport. To evaluate the sensitivity of corals to transport stress, we examined the survival of corals collected and returned to their DSCSC on the same day with a similar group held overnight in shipboard aquaria then relocated after ca. 1 day. For this experiment, only attachment of fragments by cement (coral pot v.2) was used. For the same-day treatment, multiple species were collected and processed while the corals were held in the ROV biobox to avoid stress (**Supplementary Figure 1A**). Using a team of six people with designated responsibilities, processing and attachment of each coral fragment required an average of 7 min per fragment (n = 20 coral fragments of various taxa). Once all fragments were fixed in coral pots, the ROV was deployed to relocate the corals at their collection location on Sur Ridge. This same-day approach required ca. 2 h at the surface and 1 h to return the coral pots to depth. For the overnight treatment, corals were maintained in shipboard aquaria overnight using 50-gallon acrylic aquaria (n = 3) in a darkened, temperature-controlled room. Chilled seawater (5–6°C) was set to flow at 1 $Lmin^{-1}$ and the lights were turned off except during periods when corals were processed (**Supplementary Figure 1B**). Donor coral branches were constantly immersed in large Tupperware containers during movement from the ROV biobox to shipboard aquaria—ensuring the corals would not be exposed to air. Once in aquaria, the collected corals were fragmented and attached to coral pots (v.2). On the day after collection and processing (approximately 12–18 h post collection), the coral pots were transferred to the ROV biobox in large Tupperware containers for translocation to the study sites. The biobox was prepared to receive the coral pots by filling the partitions with the same shipboard chilled seawater used for the aquaria and with ice packs to maintain seawater temperature similar to their collection location.

Coral Survivorship and Analysis

Coral translocation sites were revisited one or more times per year for two plus years to determine the survival of translocated coral fragments. Corals were determined to be alive if the observed fragments were present in the coral pot at the time of the census and exhibited normal polyp and tissue color. After approximately 3 years of repeated visits to the study sites, overall survivorship patterns are reported by attachment method and by transport stress treatment. Survivorship over time was statistically analyzed using the generalized linear model (GLM) and segmented model with the deployment group and duration of transport stress as a combined fixed factor due to the low number of replicates. Segmented model analysis was used to test for any temporal sensitivity in survivorship by group deployment and transport stress—i.e., non-linear survival vs. time (Muggeo, 2003, 2008). Model fits were compared using the corrected Akaike Information Criterion (AICc). Reduction of the AICc by more than 10 by the segmented model was assumed to be a conservative indicator of a break-point in survival. Statistics were performed using the software R and the packages "lme4," "segmented," and "AICcmodavg." We also report the survivorship patterns of each coral taxon as points for further discussion.

RESULTS

Environmental Variability and Exposure

Temperature, salinity, and dissolved oxygen data logged from 0 to 1400 m depth during the coral translocation studies are shown in **Supplementary Figures 2A–D** (n = 14 dives). During these ROV dives, the corals were generally exposed to depths of 940.13 ± 344 m (mean ± SD), temperatures of 4.4 ± 2.18°C, salinity of 34.39 ± 0.22 PSU, and dissolved oxygen levels of 0.73 ± 0.91 ml/l (**Table 2**). Temperature sensors on the ROV and within the ROV biobox indicate that corals in the biobox experienced approximately 3°C warming during the ascent and descent of the ROV. Temperatures in the biobox rose from 5.2 to 8.5°C during ROV decent over the first 30 min (**Supplementary Figure 3**). Subsequently, temperatures within the biobox dropped to ambient (4.5°C) over the following 45 min. For corals held overnight in shipboard aquaria, seawater temperature (digital handheld thermometer; n = 12 samples) was stable (5.73 ± 0.42°C) and approximately 1.3°C warmer than the bottom temperatures at Sur Ridge.

Coral Survivorship by Attachment and Temporal Sensitivity

Survival of coral fragments seemed to be dependent on the type of fragment attachment method—i.e., coral pot version—and non-linear over time. All *Keratoisis* sp., *I. tentaculum,* and *P. arborea* corals loosely attached by zip ties to a transportable module (coral pot v.1) did not survive (**Figure 2A**, black triangles). In contrast, coral fragments attached using cement (coral pot v.2) survived far better than those attached by zip-ties, but exhibited variable survival over time. Furthermore, some coral fragments survived 768 days displaying upright, colorful, and healthy polyps at the time of the census (see **Figures 2C,D**; *Keratoisis* sp. examples). Survivorship for all translocated coral fragments in coral pot v.2 observed within the first 365 days was ∼52.6%. At the end of the second year—i.e., between 563 and 685 days post-translocation—the overall mean survivorship was reduced to ∼23.9%. Survivorship of all the corals in coral pot v.2 was also significantly non-linear over time (**Figure 2A** and **Table 3**, $p < 0.001$). Segmented GLM analysis revealed that mortality rates were significantly higher in the first 105 ± 4.97 days (mean ± SE; **Table 3**, breaking point estimate; $p < 0.05$). After this initial period of approximately 3 months, the results indicate that overall survivorship rates became more stable with the slope of the second segment reaching near zero (**Figure 2A** and **Table 3**).

Results of the cement attachment method (coral pot v.2) may have been partially due to a species-specific response. The precious coral *Corallium* sp. and the black coral *S. kofoidi* had the highest survivorship of all translocated coral taxa when translocated using coral pot v.2. **Figure 3A** shows an example of a *Corallium* sp. fragment with extended polyps indicating good "health" 133 days after translocation. All (100%) of the translocated *Corallium* sp. (n = 5) survived ca. one year and *S. kofoidi* exhibited high survival with 80–100% of the translocated corals surviving approximately 1–2 years (**Figures 3B,C**). Attachment of bamboo corals (*Keratoisis* sp. and *I. tentaculum*) using coral pot v.2 resulted in ∼30% and 0–50% survivorship respectively in the first year (**Figures 3E,F**). Attachment of bubble gum corals (*P. arborea* and *S. cauliflora*) resulted in the lowest survivorship over time for all corals deployed using coral pot v.2 (**Figures 3G,H**). 0–40% of *P. arborea* (**Figure 3G**, group C, E_b, and G) survived to the end of the first year. *S. cauliflora* coral fragments resulted in approximately 40% survivorship after the first year but ultimately declined to 0% over the subsequent year (**Figure 3H**, group C).

Transport Stress and Survivorship

Results from the GLM segmented model analysis indicate that proportional survivorship was not significantly dependent on coral deployment group or whether the corals had overnight

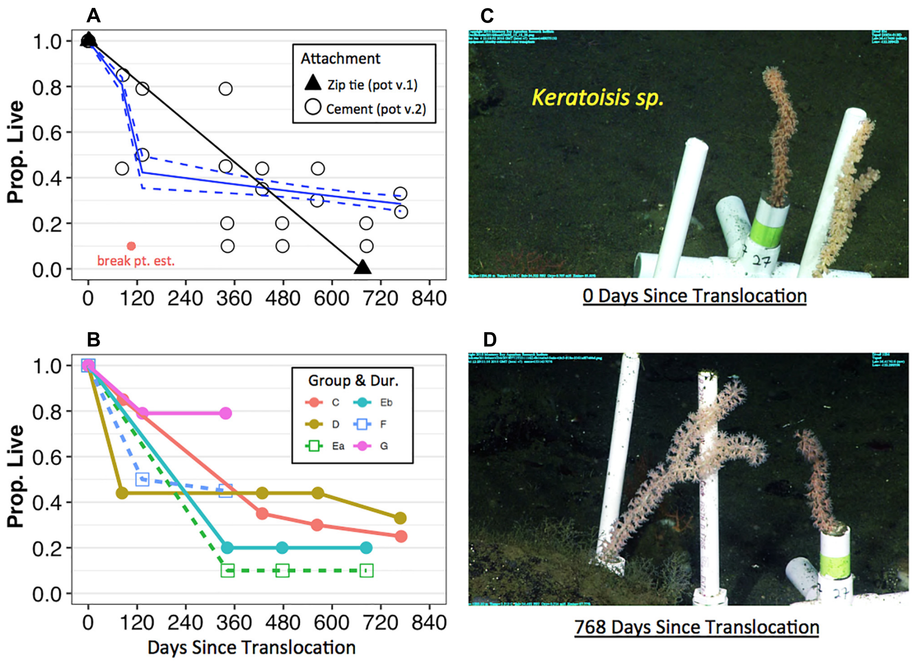

FIGURE 2 | Survivorship of translocated deep-sea corals by attachment, deployment group, and transport treatment over time at Sur Ridge, California (coral pot v.2). **(A)** Solid black triangles represent the proportion of live coral over time relative to the total number of initial fragments using the zip tie attachment method (coral pot v.1). The triangle symbols that represent data for *Keratoisis* sp., *I. tentaculum*, and *P. arborea* overlap. Open black circles represent the proportion of live corals over time using the cement attachment method (coral pot v.2). Solid red circle represents the breaking point estimate (105.5 ± 4.9 SE days) for the coral pot v.2 data. Solid blue lines indicate segmented GLM fit with upper and lower confidence limit estimates (dashed blue lines). **(B)** Solid colored circles represent the proportion of live corals relative to the total number of initial fragments since the day of deployment—i.e., these corals were translocated to depth on the same day of collection. Open square symbols represent the proportion of live corals that were translocated on the day after collection. The numbers of coral fragments are reported in **Table 1** for each deployment group and by taxa. **(C)** Example of *Keratoisis* sp. from Group D deployment—i.e., 0 days since translocation (June 4, 2016). **(D)** The same *Keratoisis* sp. samples from group D—768 days since translocation (July 12, 2018). The photo on the revisit could not be taken from exactly same camera angle but the #27 coral pot is clear in both panels **(C,D)**. For panels **(A,B)**, 0 days = the initial time when coral pots were translocated at depth.

TABLE 2 | General temperature, salinity, and dissolved oxygen conditions during ROV Doc Ricketts dives.

	n	Mean	SD	SE
Depth (m)	283062	940.13	344.45	0.65
Temperature (°C)	282985	4.48	2.18	0.00
Salinity	282917	34.39	0.22	0.00
D.O. (ml/l)	280793	0.73	0.91	0.00
Aquaria Temperature (°C)	12	5.73	0.42	0.12

D.O., dissolved oxygen; n, number of samples logged by SBE 19 CTDO; SD, standard deviation; SE, standard error. Full dive profiles are illustrated in ***Supplementary Figure 2****.*

treatment in shipboard aquaria (**Figure 2B** and **Table 3**). More specifically, while the survivorship rates of group D (same day translocation) did not significantly differ compared to the reference group C (same day translocation), the survivorship rates of groups E, F, and G did significantly differ regardless of transport duration treatment.

The responses to transport stress treatment may also have been partially due to a species-specific response. For the coral species tested for transport stress, survivorship due to treatment were mixed with the exception of *Lillipathes* sp. which resulted in 100% survivorship regardless of treatment (**Figure 3D**). For *Keratoisis* sp. and *S. cauliflora*, percent survivorship was generally lower for the coral fragments in the overnight treatment (**Figures 3E,G,H**; data denoted by open blue squares). Furthermore, survivorship of *S. cauliflora* declined more rapidly to 0% for the coral fragments in the overnight treatment (**Figure 3H**, group E_a) but this rapid decline may have been due to the limited samples ($n = 2$) with this deployment. However, survivorship of *I. tentaculum* and *P. arborea* due to transport stress treatment seemed to be mixed. For the former, 0–50% of the *I. tentaculum* [**Figure 3F**; group C ($n = 10$) and E_b ($n = 4$)] deployed survived ca. 2 years with the same day treatment but 25% survived the same amount of time when exposed to overnight treatment [**Figure 3F**; groups E_a ($n = 4$)]. For the latter coral species, corals deployed the same day treatment resulted in 0–40% survival ($n = 5$) whereas corals exposed to overnight treatment dropped to 0–20% at the end of the first year.

DISCUSSION

Restoration of deep-sea communities is a new frontier for ocean science and resource management as human activities have increasingly broad and profound effects in the deep sea. The

TABLE 3 | Segmented generalized linear model (GLM) results and breaking point estimate for coral group survivorship over time.

Segmented GLM model	y = DST + Translocation Type + U + psi + e			
	Estimate	**SE**	***z*-value**	***p*-value**
(Intercept)	7.76	1.24	6.26	***
DST	−0.08	0.01	−5.19	***
Group D (same day)	−0.14	0.14	−0.99	0.32
Group E_a (overnight)	−1.53	0.20	−7.55	***
Group E_b (same day)	−0.73	0.16	−4.57	***
Group F (overnight)	0.31	0.14	2.21	*
Group G (same day)	1.73	0.15	11.53	***
U	0.08	0.01	5.12	NA
Null deviance	2791.76 on 23 degrees of freedom			
Residual deviance	735.48 on 17 degrees of freedom			
AIC:	1482.4			
Segmented Model BP estimate	105.56 ± 4.97 SE			
	Estimate ± SE	***t*-value**	**LCI (95%)**	**UCI (95%)**
Segment 1 slope	−0.08 ± 0.01	−5.19	−0.11	−0.04
Segment 2 slope	−0.001 ± 0.0003	−3.14	−0.002	−0.0003

*Breaking point estimate and slopes of any segments are estimated using the segmented model (Muggeo, 2008). y = proportion of initial corals alive relative to the initial number of corals deployed for each group. Data from Group C corals (same day translocation) were used as the arbitrary reference. Type of transport stress treatment—i.e., same day translocation as the day of collection or kept overnight in shipboard aquaria—are indicated in parentheses. DST, days since translocation; e, error term; U, difference in slopes of the two segments; psi, breaking point estimate at each step with standard error; BP, breaking point; SE, ± standard error; NA, not applicable. LCI, lower confidence interval; UPI, upper confidence interval; *p < 0.05 significance; **p < 0.01 significance; ***p < 0.001 significance.*

effects of trawling on deep-sea coral and sponge communities were perhaps the first deep-sea ecosystem impacts to be highlighted (Koslow et al., 2000; Van Dover, 2014; Van Dover et al., 2014; Amoroso et al., 2018). We now face continuing fishing impacts, as well as host of new local to global threats ranging from deep-sea mining to climate-linked changes in ocean conditions. The 2010 BP Deepwater Horizon oil spill and its enormous scope refocused our attention on anthropogenic impacts in the deep sea. Constraining further impacts using networks of Marine Protected Areas, a successful approach in shallow marine environments (McCook et al., 2010), shows promise, but has only been applied sparingly in the deep sea (Edgar et al., 2014). Beyond protection, active restoration of damaged deep-sea assemblages may provide meaningful mitigation, but has received little to no attention—this study is one of the first to explore methods to promote the recovery of impacted deep-sea coral populations by examining the performance of multiple coral taxa.

The high initial mortality of translocated corals observed in our investigation is similar to the general patterns observed in previous restoration studies of shallow water corals (Lasker, 1990; Epstein et al., 2001; Linares et al., 2008a,b; Edwards et al., 2010; Boch and Morse, 2012). High mortality rates shortly after translocation indicate that methods minimizing translocation stress should be one of the key factors to consider in future studies in both shallow and deep-sea environments. The variation in mortality among coral taxa observed also indicates that longer-term evaluations are needed and that successful translocation of multiple coral species will require different solutions depending on taxa. Furthermore, as the effects of habitat heterogeneity on deep-sea coral restoration success remain unknown, a more rigorous investigation of how the conditions at the source colony location and how the conditions at the translocation sites may influence the survivorship of translocated corals need further attention.

We recognize that we have not fully explored the full range of factors that may affect the survivorship of translocated coral fragments. Our results indicate that the attachment of coral fragments to a substrate may be critical for coral survival and different methods employed for mitigation efforts (e.g., zip tie, cement, or other unexplored options) may influence restoration success. Previous studies in shallow coral ecosystems indicate that the size of fragments (Linares et al., 2008b), variation in currents (Jokiel, 1978; Boch and Morse, 2012), protection from predation (Baria et al., 2010; Shaver et al., 2018), and having a nursery phase before transplanting (Shafir et al., 2006), can also have a major influence on the survival of translocated corals. Reproductive condition of translocated corals may also affect their survival and potentially the period required for corals to produce and release viable gametes. In our study, we observed that some polyps of *Keratoisis* sp., *I. tentaculum,* and *Lillipathes* sp. contained eggs at the time of initial translocation. Although eggs were visible within several translocated fragments of each of these species looked to be decreasing over time, additional studies are required to determine the survivorship and reproductive contribution as a function of initial reproductive condition.

Differences in survival rates among species were somewhat surprising. Bubblegum corals, which have broad, flexible proteinaceous branches, appeared robust during handling for translocation, but had very high rates of mortality compared to other taxa. Many factors may have contributed to high

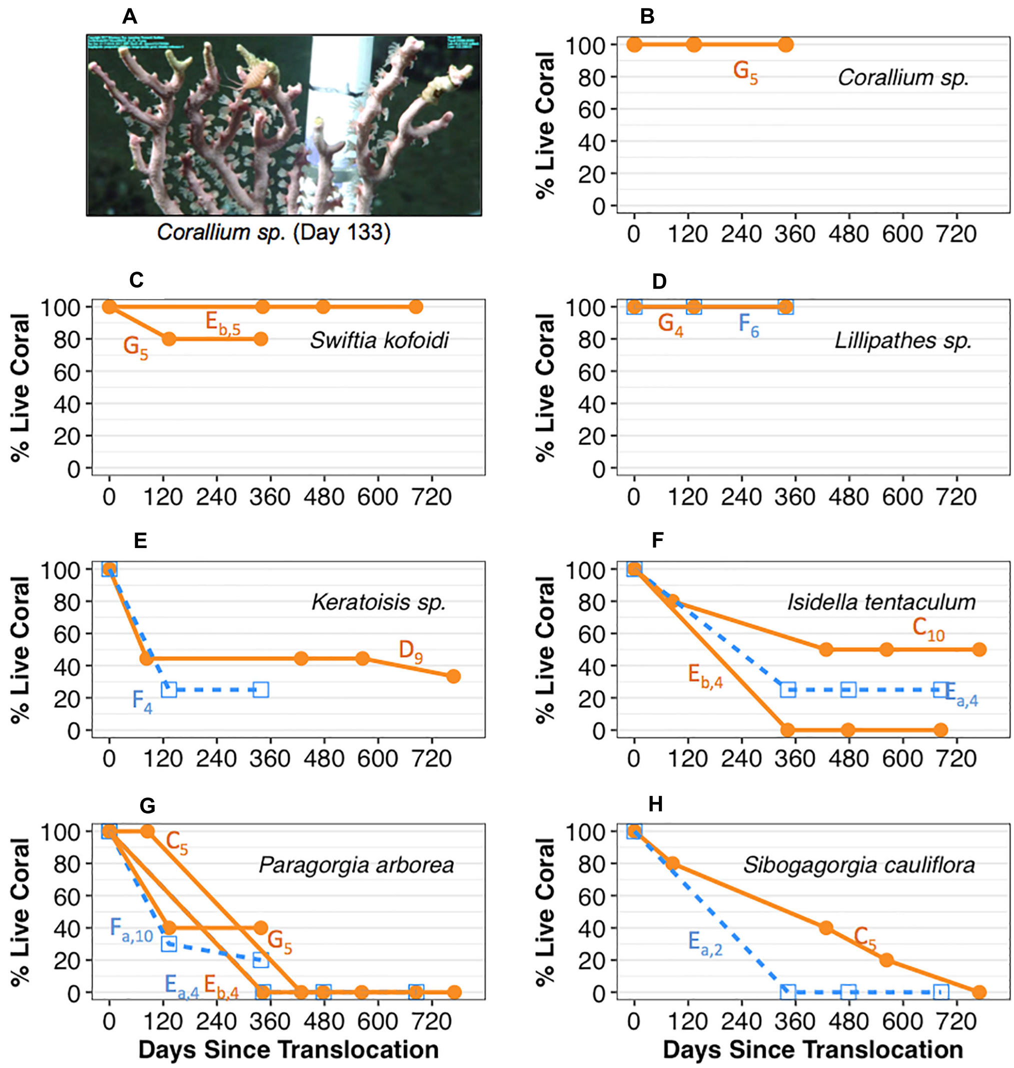

FIGURE 3 | Survivorship by coral species and transport treatment (Sur Ridge, California). **(A)** An example image of *Corallium* sp. fragment with conspicuous polyp extension at 133 days post-translocation. **(B)** Percent live *Corallium* sp. fragments over time. **(C)** Percent live *Swiftia kofoidi* fragments over time. **(D)** Percent live *Lillipathes* sp. fragments over time. **(E)** Percent live *Keratoisis* sp. fragments over time. **(F)** Percent live *Isidella tentaculum* fragments over time. **(G)** Percent live *Paragorgia arborea* fragments over time. **(H)** Percent live *Sibogagorgia cauliflora* fragments over time. For panels **(B–H)**, 0 days = the initial time when coral pots were translocated at depth. Open blue square data represent percent live of coral fragments that were exposed to overnight transport in shipboard aquaria; solid orange circle data represent survivorship for fragments that were translocated on the same day of collection. Coral group data by deployment are indicated by the letters next to each line along with the initial number of fragments at each deployment in subscript. All data represent results from deployment using the cement attachment method (coral pot v.2).

mortality rates, including the potentially toxic effects of cement or predation. While we did not have a persistent observation system to record any predatory behavior on the translocated corals, we did observe tissue sloughing from some bubble gum coral fragments (<4 months after translocation) followed by the fragments breaking off consistently at the base. Structural sensitivity of red gorgonian fragments at the base of fragments artificially attached to a natural substrate was discussed in a previous study; therefore high sensitivity at the point of attachment may be a general pattern for softer-bodied gorgonians (Linares et al., 2008a). In contrast, bamboo corals appeared to be highly fragile, but exhibited high survival up to ca. 2 year post-translocation. Thus, future studies should explore different attachment compounds and employ persistently present observation systems with a suite of sensors to help resolve questions related to factors such as attachment sensitivity and other factors such as flow and predation.

The cost of enhancing the restoration and recovery of deep-sea coral and sponge communities after anthropogenic disturbances will remain uncertain until all components of ecosystem services and the scale of active mitigation strategies can be explored. Spurgeon and Lindahl (2000) estimated that coral reef restoration costs could vary from US$13 K to US$100 M per hectare based on four case studies in shallow water coral reef systems. Additional estimates made on 10 case studies indicate that shallow water efforts have a median cost of $\sim$ US$500 K per hectare. In contrast, Van Dover et al. (2014) estimated that deep-sea restoration efforts will likely cost two to three orders of magnitude more than shallow water efforts based solely on direct costs. In our study, the use of the R/V *Western Flyer* and the ROV *Doc Ricketts* cost approximately US$30 k per day with the study site within 4 h from port. However, the cost of establishing viable translocation methods here should also include the costs of mapping the deep-sea coral and sponge communities and re-visiting mitigated areas so that outcomes of the translocation efforts can be assessed in a rigorous manner over time. Additional evaluations will also be needed to examine larger and cost-effective aquarium systems that minimize stress while transporting deep-sea corals over long distances and time periods, which may be necessary to mitigate large scale anthropogenic disturbances such as the BP Gulf of Mexico oil spill. Despite the need for further development, the relatively high survivorship of deep-sea coral fragments in pots constructed of low-cost (US$20) materials is a promising indication that developing active mitigation strategies for the DSCSCs could have merit. However, we also acknowledge that the use of PVC and cement materials is not an ideal module as a permanent transportable solution and additional bio-friendly materials will need to be explored.

The long-term survival of translocated corals, as well as their effect on DSCSC recovery over decades to centuries is yet to be determined. Considering the slow growth rates and high longevity of deep-sea corals (Andrews et al., 2002, 2005; Roark et al., 2005), it is natural to question if coral translocation will likely accelerate the recovery of damaged DSCSCs. Additionally, understanding the impacts of sourcing coral fragments from “healthy” versus “unhealthy” or “dying” donor colonies will be a critical step prior to implementation. For example, transplanting corals by fragmenting a limited number of source colonies versus fragmenting a limited number of corals that are prolific in one area to mitigate a disturbed area are likely to have different impacts on the source population but these questions have not been studied in the deep sea. Despite the gaps in knowledge, we will need to ask what role active mitigation will play in response to past, current, and future changes in the ocean due to increased human activity. Will establishing cost-effective restoration approaches that enhance gamete contact and approaches that generate corals that are more resilient to climate-related changes in ocean conditions better prepare deep-sea ecosystems for the future? Perhaps efforts to propagate other key associated taxa such as sponges that may enhance energy flow and carbon sequestration could help mitigate climate change driven changes in the deep sea (Murray et al., 1994; Cathalot et al., 2015; Kahn et al., 2015). Overall, the most effective strategies for mitigating damage in DSCSCs are uncertain but exploring the potential value of restoration options such as coral translocation and other approaches will help shape our efforts to protect and sustain these valuable and fragile deep-sea resources.

AUTHOR CONTRIBUTIONS

CB, ADV, and JB designed the study. CB did the statistical analysis. CB, JB, ADV, EB, CK, JL, and CL conducted the translocation experiments. All authors listed made a substantial contribution to the writing of the manuscript.

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The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00540/full#supplementary-material

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POLICY AND PRACTICE REVIEWS
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Voluntary Restoration: Mitigation's Silent Partner in the Quest to Reverse Coastal Wetland Loss in the USA

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Coastal ecosystems are under pressure from a vast array of anthropogenic stressors, including development and climate change, resulting in significant habitat losses globally. Conservation policies are often implemented with the intent of reducing habitat loss. However, losses already incurred will require restoration if ecosystem functions and services are to be recovered. The United States has a long history of wetland loss and recognizes that averting loss requires a multi-pronged approach including mitigation for regulated activities and non-mitigation (voluntary herein) restoration. The 1989 "No Net Loss" (NNL) policy stated the Federal government's intent that losses of wetlands would be offset by at least as many gains of wetlands. However, coastal wetlands losses result from both regulated and non-regulated activities. We examined the effectiveness of Federally funded, voluntary restoration efforts in helping avert losses of coastal wetlands by assessing: (1) What are the current and past trends in coastal wetland change in the U.S.?; and (2) How much and where are voluntary restoration efforts occurring? First, we calculated palustrine and estuarine wetland change in U.S. coastal shoreline counties using data from NOAA's Coastal Change Analysis Program, which integrates both types of potential losses and gains. We then synthesized available data on Federally funded, voluntary restoration of coastal wetlands. We found that from 1996 to 2010, the U.S. lost 139,552 acres (~565 km^2) of estuarine wetlands (2.5% of 1996 area) and 336,922 acres (~1,363 km^2) of palustrine wetlands (1.4%). From 2006 to 2015, restoration of 145,442 acres (~589 km^2) of estuarine wetlands and 154,772 acres (~626 km^2) of palustrine wetlands occurred. Further, wetland losses and restoration were not always geographically aligned, resulting in local and regional "winners" and "losers." While these restoration efforts have been considerable, restoration and mitigation collectively have not been able to keep pace with wetland losses; thus, reversing this trend will likely require greater investment in coastal habitat conservation and restoration efforts. We further conclude that "area restored," the most prevalent metric used to assess progress, is inadequate, as it does not necessarily equate to restoration of functions. Assessing the effectiveness of wetland restoration not just in the U.S., but globally, will require allocation of sufficient funding for long-term monitoring of restored wetland functions, as well as implementation of standardized methods for monitoring data collection, synthesis, interpretation, and application.

Keywords: marsh, conservation, coastal management, habitat loss, ecosystem function

INTRODUCTION

Globally, human activities have resulted in the loss of over 70% of the habitats present in 25 identified hotspots for biodiversity (Brooks et al., 2002; Ceballos et al., 2015). Habitat loss in coastal ecosystems, in particular, has been significant, with 40% to 85% of salt marshes, seagrasses, mangroves, and oyster reefs estimated to be degraded or lost regionally and globally (Kennish, 2001; Valiela et al., 2001; Waycott et al., 2009; Beck et al., 2011). Until the latter half of the twentieth century, primary anthropogenic drivers of habitat loss, such as residential and industrial development and agriculture, continued largely unchecked in most countries. Even as the proto-environmental movement became mainstream in the 1950s and early 1960s, environmental concerns largely centered on air and water pollution as a public health issue (Dewey, 1998). Increasing public awareness of environmental issues in the 1960s and 1970s, often attributed to widely publicized environmental disasters, such as oil spills, pollution in the Great Lakes, and effects of insecticide use, birthed and rapidly advanced the modern environmental movement in the United States, Europe, and elsewhere (Dryzek et al., 2003; Dunlap and Mertig, 2014). This movement directly contributed to the enactment of several environmental policies and programs, including the U.S. Clean Air Act, U.S. Clean Water Act, U.S. Endangered Species Act, the European Union Environmental Action Programmes, and the International Union for Conservation of Nature's Red List of Threatened Species. Collectively, these efforts have likely reduced the rates of habitat degradation for many critically valuable habitats regionally (Salzman, 1990; Arnold, 1991; Noss et al., 1997; Fischman, 2004). While the effectiveness of landmark environmental policies on air and water quality have been well-documented (e.g., Wooley and Wappett, 1982; Knopman and Smith, 1993; Lynch et al., 1996; Likens et al., 2001; Lyon and Stein, 2008), implementing laws and environmental programs aimed at mitigating or compensating for habitat destruction has been challenging. In the U.S., approaches to wetland protection, in particular, have evolved as policymakers and the public became increasingly aware of the causes and ecological consequences of wetland loss and degradation (Institute for Water Resources, 2018).

In 1987, the U.S. Environmental Protection Agency (EPA) convened the National Wetlands Policy Forum (NWPF), with a primary goal of addressing which policies should be adopted or amended to protect and conserve wetland resources (National Wetlands Policy Forum, 1988). The intent of the NWPF was to shift United States wetland regulation toward a policy of "No Net Loss" (NNL), specifically recommending that Federal legislation "establish a national wetlands protection policy to achieve no overall net loss of the nation's remaining wetland base, as defined by acreage and function, and to restore and create wetlands, where feasible, to increase the quality and quantity of the nation's wetland resource base" (National Wetlands Policy Forum, 1988; Bendor, 2009). The United States Army Corps of Engineers (USACE) and the EPA entered into a mitigation Memorandum of Agreement in 1990, which articulated the Clean Water Act Section 404 regulatory policy that permit applicants minimize wetland loss to the extent feasible and provide compensatory mitigation for unavoidable wetland impacts (Bendor, 2009). USACE is primarily responsible for ensuring adequate mitigation consistent with USACE and EPA regulations established under Section 404 of the Clean Water Act (Page and Wilcher, 1990; Allen and Feddema, 1996; Hough and Robertson, 2009). NNL as a policy goal, with more recent compensatory mitigation regulations, arguably continues to be a motivating force for wetlands conservation and restoration actions in the United States (Salzman and Ruhl, 2007; Bendor, 2009; Hough and Robertson, 2009).

The effectiveness of current regulations in conserving and promoting restoration of wetlands and associated ecosystem functions has been frequently questioned because of insufficient wetland mitigation following impacts, inadequate monitoring of restored wetlands to ensure recovery of function, and geographic discrepancies between where wetlands are impacted and where they are restored (Breaux and Serefiddin, 1999; Matthews and Endress, 2008; Bendor, 2009). Indeed, studies of compensatory wetland mitigation in the 1990s and early 2000s from states on the Northeast, Southeast and Pacific coastlines of the United States found less than half of permitted projects to be in compliance and approximately one in four projects did not attempt any mitigation at all (Florida Department of Environmental Regulation, 1991; DeWeese, 1994; Allen and Feddema, 1996; Brown and Veneman, 2001; Sudol and Ambrose, 2002). Further, compensatory wetlands often differ significantly in structure and function from natural reference wetlands (Balcombe et al., 2005; Spieles et al., 2006; Matthews and Endress, 2008; Hossler et al., 2011). To address these failures with compensatory mitigation, the USACE issued the "Compensatory Mitigation for Losses of Aquatic Resource Final Rule" in 2008 requiring real estate and financial instruments to protect and provide long-term stewardship of mitigation sites, respectively (Van den Bosch and Matthews, 2017). However, very few studies have addressed rates of mitigation compliance since 2008 (but see Hill et al., 2013), due in part to a major shift toward the use of in-lieu fees and mitigation banks as opposed to on-site mitigation (Hough and Harrington, 2019), and, as such, the impact of the 2008 Mitigation Rule on compliance remains unknown (Morgan and Hough, 2015).

Improved compliance with CWA Section 404 permit conditions is vital to further reduce the loss of wetland acres, to say nothing of habitat function. However, because compensation for the CWA Section 404 permitted projects is not always required or successful, voluntary restoration efforts will likely be necessary to achieve NNL. Wetland degradation and loss is also occurring as a result of natural (e.g., storm events) and anthropogenic (e.g., groundwater and oil extraction, shoreline hardening) processes that are not being prevented or mitigated under current U.S. policies and regulations (Baumann and Turner, 1990; Flournoy, 2003; Hough and Robertson, 2009; Flournoy and Fischman, 2013). Therefore, restoration efforts that compensate for wetland losses attributable to factors beyond those triggering mitigation by current U.S. laws are likely to be necessary if wetland gains are to outpace wetlands losses.

To determine how much restoration may be necessary to outpace current and future wetland losses, we first assessed how coastal wetlands have changed in the United States in the most recently reported 15 years period spanning 1996–2010, shortly after the implementation of NNL. We then synthesized data on Federally funded wetland restoration efforts that were not mandated by current U.S. compliance or mitigation requirements, termed "voluntary" herein, to determine if current efforts have the potential to outpace coastal wetland losses now and in the future. We then attempted to identify local and regional hotspots of wetland change and compare those loss/gain hotspots with restoration efforts in those areas. Further, because voluntary restoration projects were not driven by mitigation requirements that often mandate restoration to be in close spatial proximity to impacted wetlands, we investigated the degree to which siting of voluntary restoration may be creating local and regional "winners" and "losers." Finally, we make recommendations for determining how much and where future wetland restoration should occur and what additional data should be collected to inform these decisions.

APPROACH

Coastal Wetland Change in the United States

To assess how coastal wetlands have changed in the United States, we reviewed the literature and publicly available datasets on wetland coverage and trends. Only recently have technological advances allowed for national-scale assessments of wetland extent and its change over time (Davidson, 2014; NOAA, 2018a). We elected to use the data available from the National Oceanographic and Atmospheric Administration (NOAA) Coastal Change Analysis Program (C-CAP) to determine coastal wetland change since 1996, the earliest available year for which C-CAP data exist for all coastal U.S. counties. NOAA C-CAP produces nationally standardized land cover and land change data for the coastal regions of the U.S (30-m pixel resolution, based on LandSAT imagery, NOAA, 2018a). Land-cover classifications include intertidal areas, wetlands, and adjacent uplands. C-CAP calculates and publishes data on land-cover change every 5 years, with change data currently available for the 5 years periods ending in 1996, 2001, 2006, and 2010. Change in land cover over each 5 years period is determined via comparison of land-cover imagery and classification of changes in land cover using a combination of models, ancillary data, and manual edits (McCombs et al., 2016). An accuracy assessment of the change analyses from 2006 to 2010 showed an overall accuracy in classifying land cover change ranging from 82.3 percent to 85.6 percent (McCombs et al., 2016).

For this study, we used extent and change data for palustrine and estuarine wetlands from NOAA C-CAP summarized at the coastal county level over the three, 5 years periods between 1996 and 2010, as well as the full 15 years period, cumulatively. Palustrine wetlands include all non-tidal wetlands, as well as wetlands that occur in tidal areas in which salinity due to ocean-derived salts is below 0.5 psu. Estuarine wetlands include all wetlands that occur in tidal areas in which salinity due to ocean-derived salts is equal to or greater than 0.5 psu. Palustrine wetlands and estuarine wetlands were further subdivided into three subcategories: forested, scrub-shrub, and emergent (**Table 1**; NOAA, 2018b). We extracted wetland extent and change data for U.S. "coastal shoreline counties," which we define as counties that have coastlines bordering the open ocean, or contain coastal high hazard areas (V-zones, see adapted from NOAA, 2018c), to allow direct comparison to available restoration data (see *Voluntary coastal wetland restoration efforts in the United States* section below) compiled for the same coastal shoreline counties, referred to as coastal counties herein.

Voluntary Coastal Wetland Restoration Efforts in the United States

We synthesized data on voluntary coastal habitat restoration projects funded by the NOAA, EPA, the U.S. Fish and Wildlife Service (USFWS), the U.S. Department of Agriculture Natural Resource Conservation Service (USDA NRCS), and the National Fish and Wildlife Foundation (NFWF) (see **Table 2**). Projects were cross-checked across agencies to ensure that projects funded by multiple Federal sources were not double-counted in the

TABLE 1 | NOAA C-CAP wetland classifications.

	Definition
PALUSTRINE WETLANDS	
Palustrine Forested Wetland	Includes tidal and non-tidal wetlands dominated by woody vegetation ≥5 m in height, and all such wetlands that occur in tidal areas in which salinity due to ocean-derived salts is below 0.5%. Total vegetation coverage is >20%.
Palustrine Scrub/Shrub Wetland	Includes tidal and non-tidal wetlands dominated by woody vegetation <5 m in height, and all such wetlands that occur in tidal areas in which salinity due to ocean-derived salts is below 0.5%. Total vegetation coverage is >20%. Species present could be true shrubs, young trees and shrubs, or trees that are small or stunted due to environmental conditions.
Palustrine Emergent Wetland (Persistent)	Includes tidal and non-tidal wetlands dominated by persistent emergent vascular plants, emergent mosses or lichens, and all such wetlands that occur in tidal areas in which salinity due to ocean-derived salts is below 0.5%. Total vegetation cover is >80%. Plants generally remain standing until the next growing season.
ESTUARINE WETLANDS	
Estuarine Forested Wetland	Includes tidal wetlands dominated by woody vegetation ≥5 m in height, and all such wetlands that occur in tidal areas in which salinity due to ocean-derived salts is equal to or greater than 0.5%. Total vegetation coverage is >20%.
Estuarine Scrub/Shrub Wetland	Includes tidal wetlands dominated by woody vegetation <5 m in height, and all such wetlands that occur in tidal areas in which salinity due to ocean-derived salts is equal to or greater than 0.5%. Total vegetation coverage is greater than 20%.
Estuarine Emergent Wetland	Includes all tidal wetlands dominated by erect, rooted, herbaceous hydrophytes (excluding mosses and lichens). These wetlands occur in tidal areas in which salinity due to ocean-derived salts is equal to or greater than 0.5% and are present for most of the growing season in most years. Total vegetation cover is >80%. Perennial plants usually dominate these wetlands.

resulting dataset. Projects reported from each aforementioned Federal source were not solely funded by the reporting source, but were instead funded by a combination of Federal, state, and private funds via multi-entity partnerships and fund-matching requirements. Because USACE currently lacks a centralized database for voluntary restoration projects, we were unable to include those data (Vanderbilt, personal communication). Mitigation projects completed to fulfill CWA mitigation requirements or to comply with the National Resource Damage Assessment (NRDA) program were not included, as these projects are intended to mitigate or replace habitats being lost as a direct result of regulated action. We focused on voluntary restoration projects to assess the potential for these efforts to compensate for wetland losses attributable to direct human actions, as well as natural and indirect anthropogenic causes of wetland loss (e.g., storm events, sea-level rise, hydrological modification). Wetland restoration projects included in this study encompassed a wide variety of restoration techniques, including but not limited to invasive species removal, hydrologic reconnection, and wetland vegetation planting. These voluntary projects were implemented to fulfill a broad range of goals, such as improving local water quality or restoring habitat for a threatened or endangered species, depending on the mission and mandates of the Federal agency partner involved (see **Table 2** for information on restoration projects data sources).

Data availability varied across sources, with restoration projects awarded from 2006 to 2015 being available from NOAA, USFWS, USDA NRCS, and EPA's Gulf of Mexico, Chesapeake Bay and San Francisco Bay programs. Only projects awarded from 2011 to 2015 by EPA's National Estuaries Program (NEP) and NFWF were available. The habitat type, location (coastal county), and amount restored (area, in acres), were reported for each project. We then extracted all freshwater wetland, tidal wetland, and mangrove restoration projects from this larger dataset to compare to the NOAA C-CAP data. Freshwater wetlands are defined as wetlands without salt or tidal influence, including forested, scrub-shrub and emergent wetlands. Tidal wetlands were defined as forested, scrub-shrub, and emergent vegetation subjected to tidal inundation excluding wetlands dominated by mangrove species. To allow for comparison to the NOAA C-CAP data, we reclassified restoration projects as palustrine or estuarine based on the definitions described above (**Table 1**).

RESULTS

Coastal Wetland Change (1996–2010)

In 2010, there were 5,442,458 acres (~22,025 km^2) of estuarine wetlands and 23,230,861 acres (~94,012 km^2) of palustrine wetlands in the 282 conterminous coastal counties of the United States (NOAA, 2018a). A majority of extant estuarine wetlands in the U.S. was emergent tidal wetlands dominated by rooted herbaceous hydrophytes (86%), with the remainder being tidal scrub-shrub (5%) and forested (9%) wetlands. In contrast, coastal palustrine wetlands are dominated by forested wetlands (61%), with scrub-shrub and emergent wetlands making up only 15 and 24%, respectively. From 1996 to 2010, U.S. coastal counties lost 139,552 acres (~565 km^2) of estuarine wetlands (2.5% overall) and 336,922 acres (~1,363 km^2) of palustrine wetlands (1%).

Twice as much estuarine wetland area was lost in the 5-year period 2001 to 2006, as compared to the previous 5 years period from 1996 to 2000 (**Figure 1A**). From 2006 to 2010, estuarine wetland losses were nearly nine times the losses reported from 1996 to 2001 (**Figure 1A**). Ninety-one percent of estuarine wetlands losses from 1996 to 2010 were attributed to losses of emergent wetlands, with conversion to unconsolidated shoreline (loose-sediment shoreline lacking vegetation) being the primary cause of loss from 1996 to 2010 (**Figure 1A**; see NOAA, 2018b for land-cover classification definitions). At a regional level, 87% of estuarine wetlands losses occurred in coastal counties along the Gulf of Mexico (GOM); the Northeast

TABLE 2 | SNAPP restoration data sources.

Data source	Year span	URL
NOAA Restoration Center	2006–2015	https://www.fisheries.noaa.gov/topic/habitat-conservation#how-we-restore
NOAA Pacific Coast Salmon Recovery Fund	2006–2015	https://www.webapps.nwfsc.noaa.gov/
EPA National Estuary Program	2011–2015	https://www.epa.gov/nep
EPA Gulf of Mexico Program	2006–2015	https://www.epa.gov/gulfofmexico
EPA San Francisco Bay Water Quality Improvement Fund	2006–2015	https://www.epa.gov/sfbay-delta/san-francisco-bay-water-quality-improvement-fund
EPA Chesapeake Bay Program	2006–2015	https://www.epa.gov/aboutepa/about-chesapeake-bay-program-office
National Fish and Wildlife Foundation	2011–2015	https://www.nfwf.org/Pages/default.aspx
USDA Natural Resources Conservation Service	2006–2015	https://www.nrcs.usda.gov/wps/portal/nrcs/site/national/home/
USFWS Fish and Aquatic Conservation Program	2006–2015	https://www.fws.gov/fisheries/
USFWS Partners for Fish and Wildlife Program	2006–2015	https://www.fws.gov/partners/
USFWS Coastal Program	2006–2015	https://www.fws.gov/coastal/
USFWS National Wildlife Refuge System	2006–2015	https://www.fws.gov/refuges/
USFWS Wildlife and Sport Fish Restoration Program	2006–2015	https://wsfrprograms.fws.gov/

and Southeast Atlantic coast accounted for roughly 6% each, and Pacific coast accounted for <2% of all estuarine wetland loss (**Figures 2A,B**). Loss to unconsolidated shoreline was the leading cause of estuarine wetlands losses in GOM (88%), Northeast (53%), and Pacific (50%) coastal counties, while in the Southeast, loss to upland was the leading cause of estuarine wetlands loss (59%). Ninety-seven percent of estuarine wetland losses occurred in the following five states: Louisiana (80%), Florida (12%), California (2%), New Jersey (2%), and Virginia (1%) (**Figure 2**). Although Louisiana and Florida had the most estuarine wetlands to lose (54% of 1996 area), North Carolina, South Carolina and Georgia, states that also had considerable estuarine wetland area in 1996, all experienced estuarine wetland gains. North Carolina and South Carolina accounted for 79 and 18% of all wetland gains, respectively.

Palustrine wetland losses from 2001 to 2006 were nearly triple those during the previous 5 years (**Figure 1B**). During the period from 2006 to 2010, palustrine wetland losses had dropped to one third of those reported from 2001 to 2006 (**Figure 1B**). Net losses of palustrine wetlands from 1996 to 2010 represent losses of more than 1.3 million acres (~5,261 km^2) of palustrine forested wetlands, but gains of nearly 1 million acres (~4,047 km^2) scrub-shrub and emergent wetlands, often resulting from conversion of forested to scrub-shrub or emergent wetlands (**Figure 1B**). Conversion to developed lands was the greatest cause of palustrine wetland loss from 1996 to 2010 both nationally and regionally (**Figure 1B**). At a regional level, 52 and 34% of palustrine wetlands losses in coastal counties occurred along GOM and Southeast coastlines, respectively, while the Northeast and Pacific coastlines accounted for 13 and 1%, respectively (**Figures 3A,B**). Nearly 80% of palustrine wetland losses from 1996 to 2010 occurred in coastal counties within five states (listed from greatest to least loss): Florida (27%), Louisiana (17%), South Carolina (14%), Texas (12%), and North Carolina (8%, **Figure 3**). These states also had the most palustrine wetlands to lose in 1996 (71% of palustrine wetland area). California and the District of Columbia were the only state and Federal district to gain palustrine wetlands between 1996 and 2010, with California accounting for over 99% of those gains.

Voluntary Coastal Wetland Restoration 2006–2015

From 2006 to 2015, the Federal government funded the voluntary restoration of 145,443 acres (~589 km^2) of estuarine wetlands and 154,772 acres (~626 km^2) of palustrine wetlands in U.S. coastal counties. There were 748 estuarine wetland restoration projects awarded from 2006 to 2015, with an average project size (mean ± 1 standard deviation acres) of 194 ± 1,032 acres. Similarly, there were 598 palustrine wetland restoration projects awarded from 2006 to 2015, with an average project size of 259 ± 1,221 acres. Only one estuarine and one palustrine restoration project exceeded 20,000 acres (~81 km^2), while projects <1 acre accounted for 17% (129 projects) of estuarine wetlands restoration and 5% (29 projects) of palustrine wetlands restoration. On average, estuarine and palustrine wetlands restoration projects were completed within 2.16 ± 1.69 and 3.21 ± 2.10 years of being awarded, respectively. Restoration activities included, but were not limited to, vegetation planting, invasive species removal, prescribed burn, hydrologic reconnection, sediment stabilization/redistribution, and debris/pollutant removal.

More than twice as many acres of estuarine wetlands were reported as restored from 2011 to 2015 than were reported from 2006 to 2010. However, restoration projects reported by EPA NEP and NFWF accounted for 45% of the estuarine restoration that occurred between 2011 and 2015. Estuarine wetlands restoration 2011–2015 exceeded losses during each of the 5-year period between 1996 and 2010 (**Figure 1A**). By region, the Pacific coast accounted for 46% of estuarine wetland restoration acreage and 30% of the projects that occurred between 2006 and 2015, GOM counties accounted for 25% of acreage and 27% of projects, the Northeast accounted for 16% of acreage and 28% of projects, and the Southeast contributed 13% of acreage and 15% of projects. Seventy-five percent of the restored estuarine wetland acreage occurred in the following five states: California,

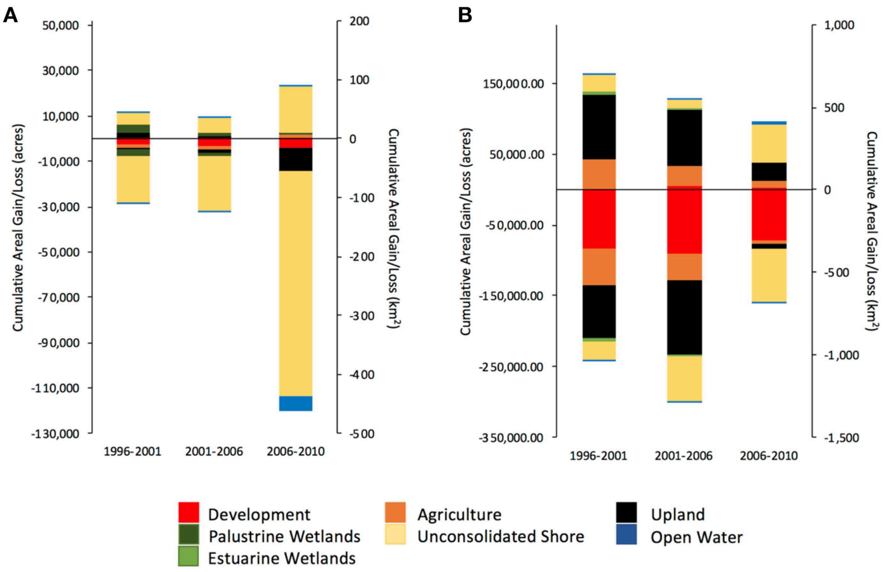

FIGURE 1 | Rate of **(A)** estuarine wetland gain/loss (acres and km^2 per year) and **(B)** palustrine wetland gain/loss in United States shoreline counties from 1996 to 2001, 2001 to 2006, and 2006 to 2010.

Texas, Delaware, Louisiana, and Washington (**Table 3**; **Figure 4**). Restoration efforts in California accounted for more than 40% of the total area restored from 2011 to 2015 and ∼40% of its 1996 estuarine wetland area (**Table 3**). Correlation analysis indicated a marginally significant positive relationship between cumulative estuarine wetlands loss between 1996 and 2010 and cumulative estuarine wetlands restoration between 2006 and 2015 at the state level (Spearman's Rank Order Correlation; $p = 0.05$, rho $= 0.43$).

The area of palustrine wetlands restored more than doubled from the first to the second half of the decade. However, palustrine restoration during the first 5-year period was less than the losses for all of the three 5-year period between 1996 and 2010 (**Figure 2B**). Restoration projects reported by EPA NEP and NFWF accounted for 82% of the palustrine restoration that occurred between 2011 and 2015; thus, they are responsible for all of the increase and compensate for what would otherwise have been a reduction in palustrine restoration effort from 2011 to 2015 compared to the prior 5-year period. Palustrine wetland restoration between 2011 and 2015 considerably exceeded losses from 1996 to 2001 and 2006 to 2010, but was only approximately half of the losses from 2001 to 2006 (**Figure 1B**). By region, coastal counties along the GOM accounted for 52% of the cumulative restored acreage of palustrine wetlands and 31% of the total number of projects that were awarded between 2006 and 2015, followed by coastal counties in the Northeast (25% of acreage, 41% of projects), the Southeast (20% of acreage, 8% of projects), and the Pacific (3% of acreage, 20% of projects). Eighty-two percent of palustrine wetland area restored from 2006 to 2015 occurred in coastal counties within five states: Florida, Maine, North Carolina, South Carolina, and Texas (**Table 4**; **Figure 5**). Restoration efforts in North Carolina accounted for ∼13% of its 1996 palustrine wetland area, the highest percentage nationally (**Table 4**). Correlation analysis indicated a significant positive relationship between cumulative palustrine wetlands loss between 1996 and 2010 and cumulative palustrine wetlands restoration between 2006 and 2015 at the state level (Spearman's Rank Order Correlation; $p = 0.02$, rho $= 0.52$).

DISCUSSION

Coastal estuarine and palustrine wetlands continue to be lost in the United States despite significant progress in achieving the goal of NNL nationally through wetland conservation and restoration efforts (NOAA, 2010; **Figure 1**). Estuarine wetland restoration efforts would likely need to more than double in order to keep pace with the recent trend of estuarine wetlands losses (2006–2010; **Figure 1A**). This target rate of restoration assumes that all voluntary wetland restoration is creating new wetlands, as opposed to sustaining or restoring existing, but degraded, estuarine wetlands. However, several of the restoration actions reported, such as debris, pollutant, and invasive species removal, are not likely to create new wetlands and thus would not contribute to offsetting wetland losses. Similarly, despite considerable voluntary efforts, the acreage of palustrine wetlands restored was insufficient to compensate for observed losses. The future potential of voluntary estuarine and palustrine wetland restoration to help offset losses will likely depend on (1) whether restoration efforts continue at the higher rate observed in the most recent 5-year period (2011–2015), as well as (2) whether major drivers of wetland loss are halted or mitigated.

Wetland mitigation requirements are designed largely to address anthropogenically induced wetland losses attributable to direct impacts of discrete events, such as filling or draining of wetlands for development or agriculture purposes (Turner, 1997; Boesch et al., 2001; Dahl, 2011). The continued and accelerating

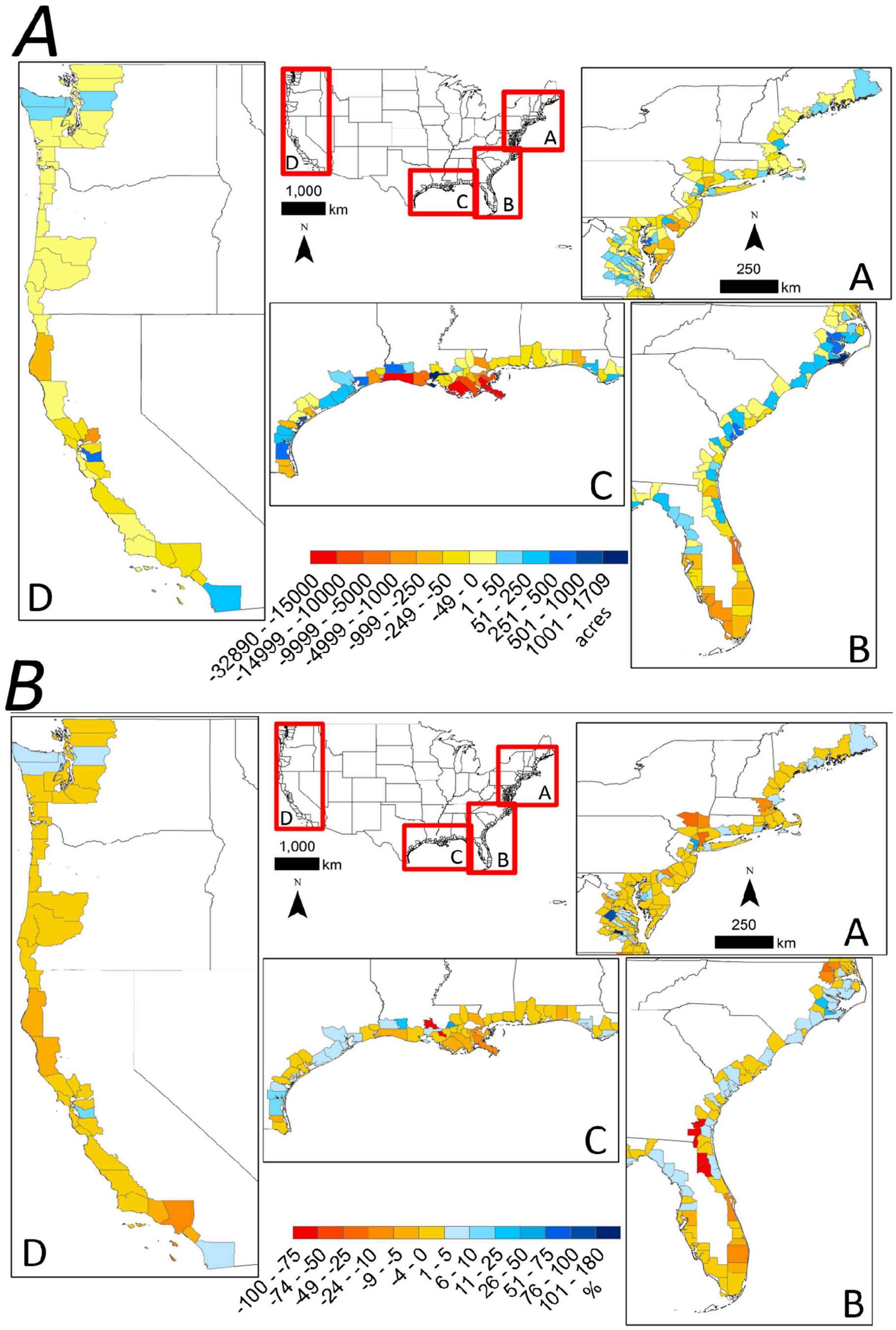

FIGURE 2 | National maps with inset panels showing coastal county-level variation in the **(A)** cumulative raw acreage change in estuarine wetlands from 1996 to 2010 and **(B)** cumulative percent change in estuarine wetlands acreage from 1996 to 2010 in the (A) Northeastern coastline, (B) Southeastern coastline and gulf coast of Florida, (C) Gulf of Mexico coastline, and (D) Pacific coastline of the conterminous United States.

losses of coastal wetlands due to direct human action may in part be explained by non-compliance with mitigation requirements. In the 1990s, inadequate compliance was a documented cause for continued wetland loss. For example, in California, only 33% of the 162 CWA permitted projects that were monitored were in compliance (DeWeese, 1994; Allen and Feddema, 1996; Sudol and Ambrose, 2002). Similarly, out of 391 projects requiring compensatory wetland loss mitigation projects in Massachusetts, 54% were not in compliance, 65% were smaller than required, and 22% did not attempt to conduct any mitigation effort (Brown and Veneman, 2001). Furthermore, in the early 1990s, Florida, the state with the most palustrine wetland loss over

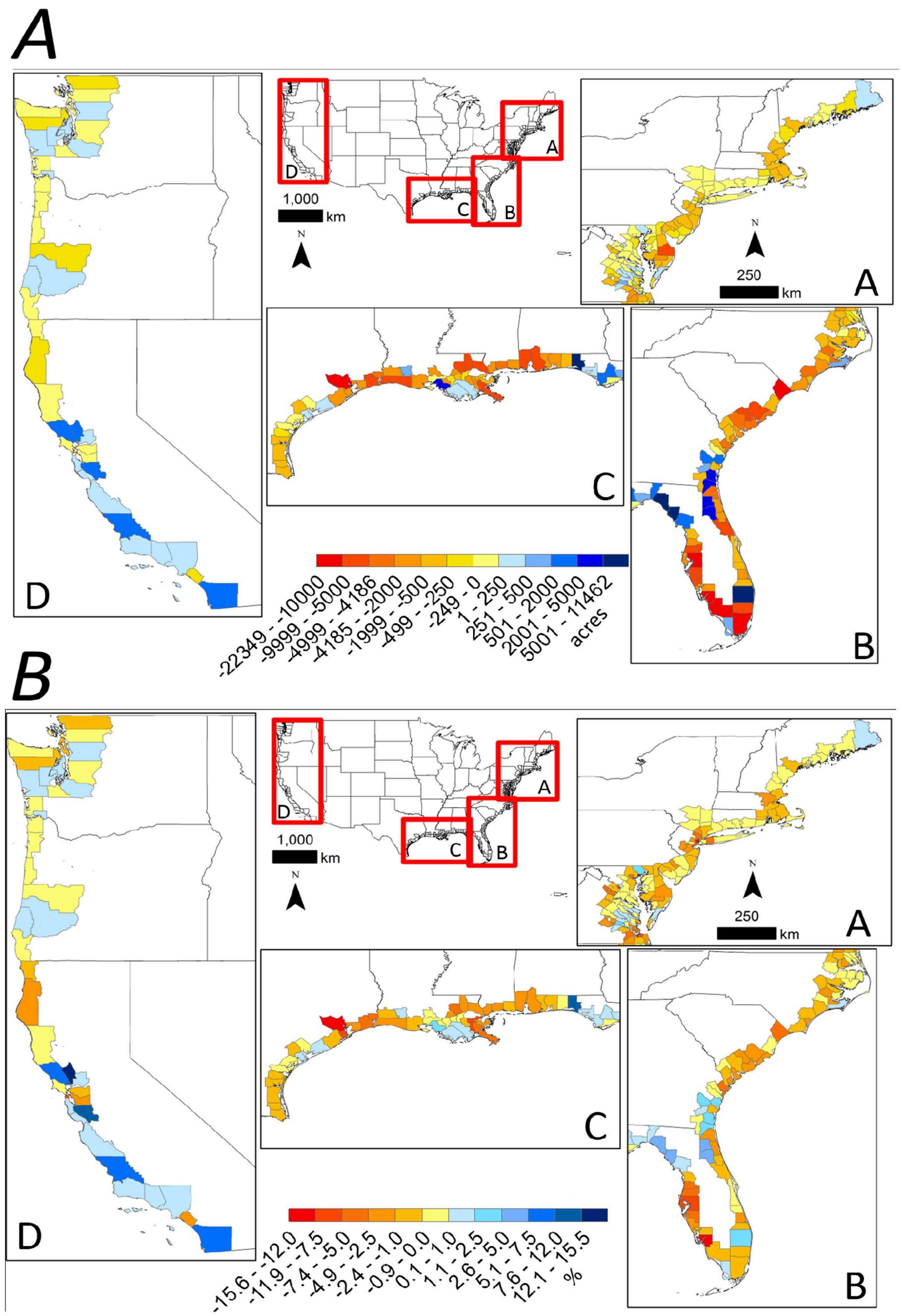

FIGURE 3 | National maps with inset panels showing coastal county-level variation in the **(A)** cumulative raw acreage change in palustrine wetlands from 1996 to 2010 and **(B)** cumulative percent change in palustrine wetlands acreage from 1996 to 2010 in the (A) Northeastern coastline, (B) Southeastern coastline and gulf coast of Florida, (C) Gulf of Mexico coastline, and (D) Pacific coastline of the conterminous United States.

our study period, reported that out of 63 freshwater wetland mitigation permits reviewed, only four were in compliance, and 34% of projects were never constructed (Florida Department of Environmental Regulation, 1991). Acknowledging the shortcomings of permittee-responsible mitigation, the U.S. Army Corp's 2008 Mitigation Rule incentivized the use of mitigation banks, the number of which more than doubled in the decade after the Mitigation Rule, as well as in-lieu fees (Hough and Harrington, 2019). In the wake of the 2008 financial crisis, permittee-responsible mitigation declined from 59% of compensatory measures in 2008 to 37.5% of measures in 2014 (Madsen et al., 2010; Institute for Water Resources, 2015), and the availability of funding to study mitigation sites was reduced. Thus, whether the 2008 Mitigation Rule has changed

TABLE 3 | Voluntary estuarine restoration efforts from 2006 to 2015.

	Projects		Acres		Square kilometers		Total restoration		1996 Wetland area	
State	**2006–2010**	**2011–2015**	**2006–2010**	**2011–2015**	**2006–2010**	**2011–2015**	**2006–2010**	**2011–2015**	**2006–2010**	**2011–2015**
AL	1	1	4	1	<1	<1	<1%	<1%	<1%	<1%
CA	39	44	7,180	44,166	29	179	17%	43%	6%	40%
CT	3	18	59	97	<1	<1	<1%	<1%	<1%	1%
DE	1	18	1,980	12,003	8	49	5%	12%	3%	16%
FL	47	80	4,208	8,107	17	33	10%	8%	<1%	1%
GA	2	3	100	51	<1	<1	<1%	<1%	<1%	<1%
LA	34	31	8,732	4,707	35	19	21%	5%	<1%	<1%
MA	14	17	238	1,160	1	5	1%	1%	1%	5%
MD	25	22	118	317	<1	1	<1%	<1%	<1%	<1%
ME	5	5	247	65	1	<1	1%	<1%	<1%	<1%
MS	7	5	1,666	1,829	7	7	4%	2%	3%	3%
NC	9	12	4	5,063	<1	20	<1%	5%	<1%	2%
NH	4	1	126	<1	1	<1	<1%	<1%	2%	0%
NJ	5	16	203	3,564	1	14	<1%	3%	<1%	2%
NY	9	18	577	600	2	2	1%	1%	1%	1%
OR	14	20	1,293	1,542	5	6	3%	1%	10%	12%
RI	5	2	48	112	<1	<1	<1%	<1%	1%	1%
SC	18	9	3,578	252	14	1	9%	<1%	1%	<1%
TX	40	15	6,818	9,783	28	40	16%	9%	1%	2%
VA	8	15	52	1,470	<1	6	<1%	1%	<1%	1%
WA	53	53	4,629	8,695	19	35	11%	8%	24%	44%

The number of projects, acres, and square kilometers restored, as well as the percentage of the total national wetland area restored and the percentage of the 1996 wetland area restored are reported for coastal shoreline counties in each U.S. state for 2006–2010 and for 2011–2015.

the long-term outcomes of compensatory wetland restoration or creation remains to be seen (but see Hill et al., 2013; Van den Bosch and Matthews, 2017).

While an updated evaluation of permit compliance is needed, even among completed mitigation projects, there is considerable evidence that restored wetlands do not perform ecosystem functions equivalent to those of undisturbed wetlands (Turner et al., 2001; Gutrich and Hitzhusen, 2004; Moreno-Mateos et al., 2012). Further, increased reliance on mitigation bank credits, while they in some cases may achieve greater compliance than reported in studies from the 1990s and early 2000s, is not without risks (see Levrel et al., 2017). Mitigation banking can facilitate development as opposed to avoidance (Walker et al., 2009), result in the homogenization of wetlands due to market forces (Walker et al., 2009; Dauguet, 2015), increase the spatial disconnect between impact sites and compensatory wetland creation (Ruhl and Salzman, 2006; Bendor and Riggsbee, 2011), and reduce the likelihood of long-term monitoring due to bankruptcy (Gardner and Pulley Radwan, 2005; Robertson, 2008). As such, our results revealing that 10 years of palustrine wetlands restoration efforts (2006–2015) would have more than compensated for 15 years of losses (1996–2010) had there been no loss to development highlight the need for improved monitoring of wetland mitigation projects and increased scrutiny of un-regulated impacts of development on wetlands. These changes will be particularly critical to palustrine wetland conservation as predicted increases in coastal population density will likely result in continued development of coastal lands (NOAA, 2015).

Although wetland losses in many areas may be attributed to discrete or direct human actions, such as draining or filling for development agriculture, there are numerous natural and anthropogenic factors that can indirectly contribute to wetland losses. Secondary outcomes of increasing conversion to unconsolidated shore and open water are caused by natural processes, such as storms and flooding, as well as human activities, such as groundwater and oil extraction, installation of hydrologic barriers, sediment restriction, dredging, and climate change (Baumann and Turner, 1990; Turner, 1997; Brinson and Malvárez, 2002; Zedler and Kercher, 2005; Gittman et al., 2015; NOAA, 2018a). A major weakness of current U.S. habitat protection policies is that they are poorly suited to address indirect causes of wetland loss (Flournoy and Fischman, 2013). Thus, the degree to which wetland losses can be outpaced may depend on voluntary wetland restoration efforts.

Given that funding for wetland restoration in the U.S. and elsewhere is likely to remain limited in the future, policymakers and restoration practitioners must avoid wetland loss and prioritize where and how to regain lost coastal wetlands. Our results suggest that there are mismatches between regions where wetlands are being lost and where restoration efforts are occurring, with the greatest mismatch occurring in Louisiana, where considerably more wetlands are

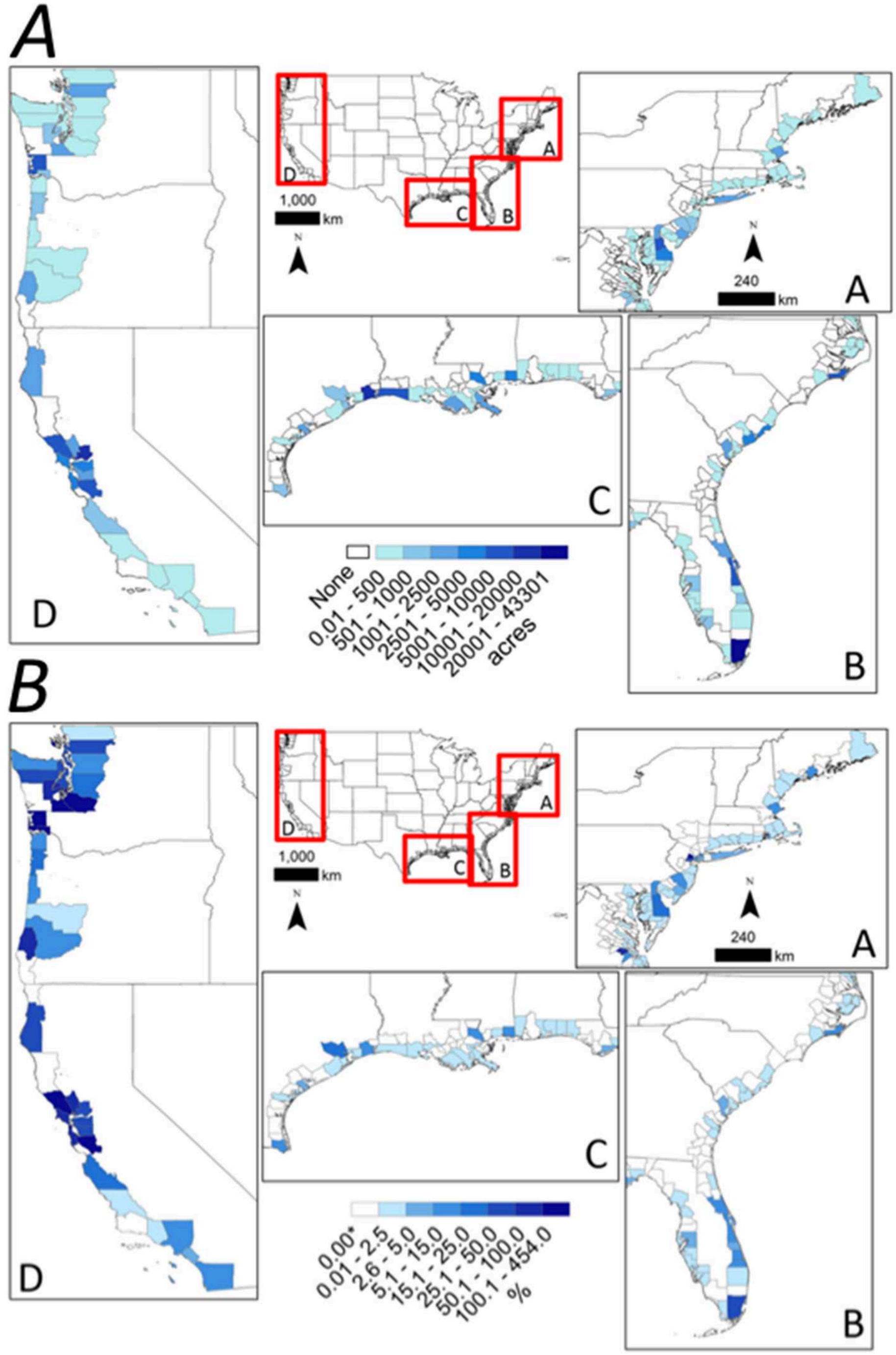

FIGURE 4 | National maps with inset panels showing coastal county-level variation in voluntary estuarine wetland restoration efforts [cumulative acreage **(A)** and % of 1996 estuarine wetland area **(B)**] along the (A) Northeastern coastline, (B) Southeastern coastline and gulf coast of Florida, (C) Gulf of Mexico coastline, and (D) Pacific coastline of the conterminous United States.

being lost than restored (**Tables 3**, **4**). However, in 2012 and updated in 2017, Louisiana adopted a Coastal Master Plan intended to direct resources to reverse the State's wetland loss over the next 50 years (CPRA, 2017). Further, this mismatch may be diminished in the coming decades, as billions of dollars have been allocated to habitat restoration in the GOM coast as a result of settlement dollars from the Deepwater Horizon oil spill in 2010 (Diamond et al., 2014). Efforts to rectify spatial mismatches between wetland loss and restoration will potentially enhance the efficacy of

TABLE 4 | Voluntary palustrine restoration efforts from 2006 to 2015.

	Projects		Acres		Square kilometers		Total restoration		1996 Wetland area	
State	**2006–2010**	**2011–2015**	**2006–2010**	**2011–2015**	**2006–2010**	**2011–2015**	**2006–2010**	**2011–2015**	**2006–2010**	**2011–2015**
AL	2	12	144	562	1	2	<1%	1%	0%	0%
CA	13	4	466	26	2	<1	1%	<1%	<1%	<1%
CT	0	2	0	2	0	<1	0%	<1%	0%	<1%
DE	25	5	1,239	94	5	<1	3%	<1%	1%	<1%
FL	32	118	4,589	69,856	19	283	10%	66%	0%	1%
GA	5	4	1,879	1,434	8	6	4%	1%	<1%	<1%
LA	3	9	97	635	<1	3	<1%	1%	<1%	<1%
MA	15	6	322	398	1	2	1%	<1%	<1%	<1%
MD	22	22	3,702	581	15	2	8%	1%	1%	<1%
ME	7	7	13,841	5,851	56	24	29%	5%	4%	2%
MS	0	0	0	0	0	0	0%	0%	0%	0%
NC	1	14	380	12,608	2	51	1%	12%	<1%	13%
NH	65	13	1,906	290	8	1	4%	<1%	<1%	<1%
NJ	5	12	754	166	3	1	2%	<1%	<1%	<1%
NY	21	5	1,174	100	5	<1	2%	<1%	<1%	<1%
OR	30	29	409	990	2	4	1%	1%	<1%	<1%
RI	1	0	1,426	0	6	0	3%	0%	2%	0%
SC	8	5	9,601	2,887	39	12	20%	3%	1%	<1%
TX	8	12	1,419	5,543	6	22	3%	5%	<1%	<1%
VA	4	6	1,772	4,254	7	17	4%	4%	<1%	<1%
WA	34	12	3,003	373	12	2	6%	0%	1%	<1%

The number of projects, acres, and square kilometers restored, as well as the percentage of the total national wetland area restored and the percentage of the 1996 wetland area restored are reported for coastal shoreline counties in each U.S. state for 2006–2010 and for 2011–2015.

restoration and minimize future disparities that will likely occur (Elliott et al., 2019).

There is mounting evidence that anthropogenic climate change effects will not be uniformly distributed along coastlines throughout the U.S. (Weston, 2014; Schuerch et al., 2018). As climate change results in sea level rise and increased storm frequency and intensity, rates of estuarine and palustrine wetlands losses are likely to accelerate, particularly in areas with highly developed uplands and sediment deficits that prevent wetlands from either transgressing landward (i.e., coastal squeeze) or accreting fast enough to keep pace with sea-level rise (Boesch et al., 2001; Scavia et al., 2002; Nicholls and Lowe, 2004; Pontee, 2013; Weston, 2014; Peteet et al., 2018; Schuerch et al., 2018). Additionally, accelerating rates of sea level rise and associated saltwater intrusion will likely result in conversion of palustrine wetlands to estuarine wetlands, unconsolidated shore, or open water, resulting in further losses (Sallenger et al., 2012; Neubauer, 2013; Peterson and Li, 2015; Valle-Levinson et al., 2017). Thus, dedicating restoration resources to areas experiencing the greatest losses may be suboptimal if local conditions make successful restoration unlikely to be achieved and sustained. As such, allocating restoration funding to wetland construction projects in regions where human activities have negatively impacted the ecogeomorphic feedbacks that support marsh stability (e.g., flood control levees in Louisiana, canal creation in Florida) may be futile without first removing the underlying indirect causes of wetland instability (Day et al., 2005; Kirwan et al., 2010; Kirwan and Megonigal, 2013; Weston, 2014; Temmerman and Kirwan, 2015).

Interpretation of our results must be caveated with an acknowledgment that our dataset did not capture all voluntary wetland restoration (e.g., EPA NEP, and NFWF projects between 2006 and 2010, USACE state agency and NGO projects without Federal-funding partners). As stated previously, not all voluntary wetland restoration is creating new wetlands, as several of the restoration actions reported (e.g., debris, pollutant, and invasive species removal) are not creating new wetlands. Further, successful wetland restoration projects can require upwards of a decade to vegetate (Zedler and Callaway, 1999; Zedler and Kercher, 2005; Kusler, 2012), resulting in a lag in the spectral change required for detection via the remote sensing approach (Chapple and Dronova, 2017) used by NOAA C-CAP to calculate wetland change (NOAA, 2018a). Thus, the potential lag in detectability of both compensatory and voluntary wetland restoration may result in an over or under estimation of wetland losses and gains for decades or even longer. While beyond the scope of the present study, advancements in the remote sensing of fine-scale land cover changes will likely become an increasingly important tool used to inform the outcome of wetland restoration, both compensatory and voluntary, at a national scale. Despite these data limitations,

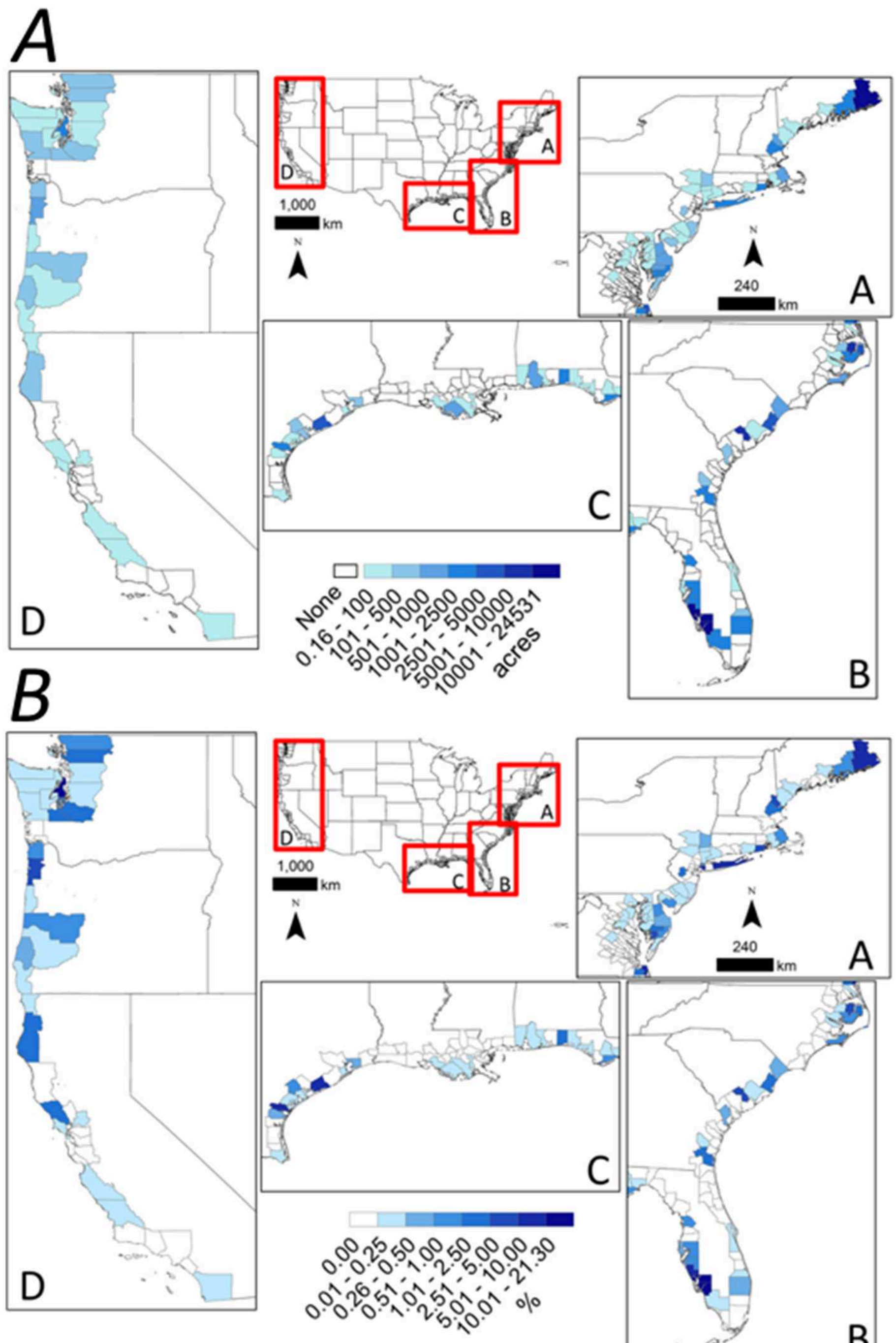

FIGURE 5 | National maps with inset panels showing coastal county-level variation in voluntary palustrine wetland restoration efforts [cumulative acreage **(A)** and % of 1996 palustrine wetland area **(B)**] along the (A) Northeastern coastline, (B) Southeastern coastline and gulf coast of Florida, (C) Gulf of Mexico coastline, and (D) Pacific coastline of the conterminous United States.

the vast majority of voluntary restoration projects included in this study were awarded to state agencies and NGOs or included state or NGO partners who generally contribute a minimum of 1:1 matching funds or services. This suggests that the Federal government is a primary catalyst for funding sources of coastal wetland restoration, and that the Federal government is likely involved in much of the voluntary restoration occurring in U.S.

We recommend the following actions for improving wetland conservation and restoration in the U.S. and globally:

- Where possible, prioritize voluntary restoration efforts in the areas that have experienced the greatest losses, while also considering local and regional natural and anthropogenic factors that may influence long-term wetland restoration success;
- Establish uniform performance metrics and monitoring protocols for assessing ecosystem functions of restored wetlands;
- Ensure adequate funding for post-restoration monitoring of created and enhanced wetlands; and
- Adopt uniform reporting practices for wetland restoration projects across restoration funders and practitioners.

Most restoration projects do not have funding for long-term monitoring post-restoration (Sutton-Grier et al., 2018); thus, long-term assessments of restored wetland resilience are rare (but see Craft et al., 2008). Further, "area restored" is the most consistently reported metric for restoration projects, yet this metric provides no information on the success of restoring ecological functions and associated services. Without the ability to determine the degree to which restored wetlands are recovering ecosystem functions equivalent to those of undisturbed wetlands, restoration cannot be completely relied upon as an effective approach to wetland protection and conservation. Policymakers and practitioners should look to recent efforts to standardize monitoring of oyster reef restoration (Brumbaugh et al., 2006; Baggett et al., 2015), as well as evaluations of restored wetland ecosystem functions (Meli et al., 2014), for further guidance. In conclusion, the results of this study suggest that reversing coastal wetland losses will be challenging to achieve as climate change exacerbates wetland loss. However, given the magnitude of recent restoration efforts, it is clear that significantly increased funding and appropriate planning and siting of coastal wetland restoration has the potential to ensure that coastal wetlands and their associated ecosystem services are protected and sustained in the future.

DATA AVAILABILITY

The datasets generated for this study are available on request to the corresponding author.

AUTHOR CONTRIBUTIONS

All authors conceived and outlined the framework for this research as part of a Science for Nature and People Partnership (SNAPP) Coastal Restoration Working Group, led by JG, KA, BD, and RG. RG and CB wrote the initial draft of the paper. All authors contributed to writing, editing, and revising subsequent drafts of the paper. RG extracted the NOAA C-CAP data with the assistance of NH. RG, CB, and JG synthesized the voluntary restoration data with the assistance of RB, JB, ACh, RHou, RHow, and TS. RG and CB analyzed the data and created the figures.

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Advancing Coastal Risk Reduction Science and Implementation by Accounting for Climate, Ecosystems, and People

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Climate change and population growth are degrading coastal ecosystems and increasing risks to communities and infrastructure. Reliance on seawalls and other types of hardened shorelines is unsustainable in an era of rising seas, given the costs to build and maintain these structures and their unintended consequences on ecosystems. This is especially true for communities that depend on coastal and marine ecosystems for livelihoods and sustenance. Protecting and restoring coral reefs and coastal forests can be lower cost, sustainable alternatives for shoreline protection. However, decision-makers often lack basic information about where and under what conditions ecosystems reduce risk to coastal hazards and who would benefit. To better understand where to prioritize ecosystems for coastal protection, we assessed risk reduction provided by coral reefs, mangroves, and seagrass along the entire coast of The Bahamas, under current and future climate scenarios. Modeled results show that the population most exposed to coastal hazards would more than double with future sea-level rise and more than triple if ecosystems were lost or degraded. We also found that ecosystem-based risk reduction differs across islands due to variation in a suite of ecological, physical, and social variables. On some populated islands, like Grand Bahama and Abaco, habitats provide protection to disproportionately large numbers of people compared to the rest of the country. Risk reduction provided by ecosystems is also evident for several sparsely populated, remote coastal communities, which in some cases, have large elderly populations. The results from our analyses were critical for engaging policy-makers in discussions about employing natural and nature-based features for coastal resilience. After hurricanes Joaquin and Matthew hit The Bahamas in 2016 and 2017, our assessment of coastal risk reduction and the multiple benefits provided by coastal ecosystems helped pave the way for an innovative loan from the Inter-American Development Bank to the Government of The Bahamas to invest in mangrove restoration for coastal resilience. This work serves as an example for other regions and investors aiming to use assessments of ecosystem services to inform financing of natural and nature-based approaches for coastal resilience and climate adaptation.

Keywords: coastal protection, coastal habitats, coastal hazards, ecosystem services, social vulnerability, sea level rise, The Bahamas, natural and nature-based features

INTRODUCTION

Coastal areas are hazard prone. An estimated 310 million people and $11 trillion in GDP are exposed globally to the extent of a 100-year flood event (Hinkel et al., 2014). Risk is expected only to increase, due to rising sea levels and other climate-related threats concurrent with population growth. By 2060, up to 411 million people could be exposed to a 100-year flood event (Hallegatte et al., 2013; Wong et al., 2014; Neumann et al., 2015; Reguero et al., 2015). Thus, building resilient communities is a shared challenge for the world's population living along the coast now and in the future (Adger et al., 2005; McGranahan et al., 2007; Kron, 2013). To address this challenge, communities typically engineer barriers along the coast. However, there is growing understanding that traditional approaches to coastal protection (e.g., seawalls, bulkheads, etc.) are unsustainable. Hardened shorelines can be expensive to build and maintain, and can lead to unintended shoreline erosion, degradation or loss of habitat, and impacts on communities that depend on healthy coastal ecosystems for protection, subsistence, and livelihoods (Burgess et al., 2004; Hillen et al., 2010; Jones et al., 2012; Gittman et al., 2015; Rangel-Buitrago et al., 2017).

Concerns about hardened shorelines are heightening interest in alternatives for coastal protection that may be less environmentally destructive, more cost-effective to maintain in the long-term, and able to provide valuable co-benefits such as habitat for fisheries (Cheong et al., 2013; Mycoo and Donovan, 2017; Reguero et al., 2018b). Coastal habitats like coral and oyster reefs, seagrass beds, marshes, mangrove, and coastal forests have the potential to attenuate waves and surge associated with storms, in some cases mitigating flooding and coastal erosion (e.g., Danielsen et al., 2005; Alongi, 2008; Barbier et al., 2008; Das and Vincent, 2009; Zhang et al., 2012; Arkema et al., 2013; Ferrario et al., 2014; Spalding et al., 2014; Narayan et al., 2016; Beck et al., 2018; Reguero et al., 2018a,b). Because conserving or restoring natural habitats does not preclude alternative actions later, nature-based approaches are generally seen as potentially no- or low-regret coastal adaptation options, irrespective of future climate (Cheong et al., 2013; Nurse et al., 2014). Conservation and in some cases, habitat restoration, is cheaper than built infrastructure, and has been found to be highly cost effective (i.e., mangrove restoration vs. breakwater construction) (Narayan et al., 2016). Furthermore, coastal ecosystems may adapt to climate change, potentially making them more effective in the long term compared to hard infrastructure (Temmerman et al., 2013). These findings are promising for island nations throughout the Caribbean, with large highly exposed coastal zones and extensive networks of ecosystems that provide multiple lines of defense against coastal hazards (Guannel et al., 2016; Beck et al., 2018). However, the ability of habitats to provide coastal protection is highly context dependent, and ecosystems perform differently based on the conditions (Costanza et al., 2008; Ruckelshaus et al., 2016; Arkema K. K. et al., 2017). Strengthening hazard resilience requires a better understanding of where ecosystems are most important for providing coastal protection, especially given threats from development (Gittman et al., 2015).

The need to better understand risk from coastal hazards and bolster shoreline resilience was brought into stark relief following the devastation from the 2017 Atlantic hurricane season, which recorded three of the top five costliest hurricanes for this region in history – Hurricanes Harvey, Maria, and Irma (National Oceanic and Atmospheric Administration [NOAA] and National Hurricane Center [NHC], 2018; EM-DAT International Disaster Database, 2018). Recovery from the 2017 season was especially protracted in the Caribbean, which highlights costly inequities in vulnerability and resilience among nations, and underscores the disproportionate burden that small island developing states will bear in adapting to climate change (Anthoff et al., 2010; Nurse et al., 2014; Wong et al., 2014; Mycoo and Donovan, 2017; Beck et al., 2018). In contrast to the mainland United States where basic services were restored within days, many residents throughout the Caribbean were without basic services for months, even up a year in the case of Hurricane Maria and Puerto Rico (e.g., Hincks, 2017; Kishore et al., 2018; Puerto Rico: The Forgotten Island, 2018; Shultz et al., 2018)[1]. Loss of life also varied dramatically, from almost 3,000 deaths on Puerto Rico, a poor island in the Caribbean, vs. four on the United States mainland (Ascertainment of the estimated excess mortality from Hurricane Maria in Puerto Rico, 2018; Pasch et al., 2019).

Variation in impacts from storms and the pace of post-disaster reconstruction highlights some of the challenges faced by island nations, including: high exposure to hazards, geographic isolation and small size, fragile infrastructure grids, and poor home construction (Ghosal, 2016; Panditharante, 2018; Rodríguez-Díaz, 2018; Shultz et al., 2018). The ability to respond to and recover from disasters is often highly variable and attention is increasingly being paid to social risk factors that make certain communities especially vulnerable to hazards. There is general consensus that factors such as age, race/ethnicity, gender, education and poverty status are quantifiable indicators for differences in access to resources, power and capacity that underlie social vulnerability (e.g., Cutter et al., 2003; Cutter et al., 2009; Peacock et al., 2012; Wamsley et al., 2015;

[1] http://status.pr/ (Retrieved Febuary 13, 2018).

Arkema K. K. et al., 2017). However, while a growing number of studies are beginning to use these demographic metrics to map risk to vulnerable communities (Boruff et al., 2005; Thatcher, 2013; Koks et al., 2015), less attention has been paid to the relationship between vulnerable communities and coastal ecosystems that may be providing risk reduction (Arkema K. K. et al., 2017).

To stimulate widespread uptake and implementation of nature-based coastal protection strategies, decision-makers also need approaches and tools that can synthesize physical, demographic and ecological data to identify in a spatially explicit manner where ecosystems matter to vulnerable communities, and evaluate alternatives in a timely manner. Benefits from the protection service provided by coastal habitats have been measured in a variety of ways. These include employing process-based predictive modeling using expected damage functions to assist in cost-benefit analysis for protective interventions (this approach is used by U.S. Federal Emergency Management Agency's HAZUS software) (Barbier, 2015). Other approaches include measuring protective benefits as capitalized into housing values using hedonic analysis (Dundas, 2017), asking people their willingness to pay for protection services using stated preference surveys (Landry et al., 2011), or using basic regression analysis as a means of relating the presence of coastal habitats to reduced flood damage (Danielsen et al., 2005; Costanza et al., 2008; Das and Vincent, 2009; Boutwell and Westra, 2016). Although all of these approaches can be used for decision support, they are data intensive, generally relying on existing data on the physical drivers of storm risk, geospatial information on exposed people and infrastructure, or extensive primary data collection. Data requirements notwithstanding, these approaches may also require significant expertise to run (i.e., complex wave models may take months to parameterize by a coastal engineer), which can make it more difficult for staff in organizations with limited capacity to quickly iterate scenarios and consider in quantitative terms the competing goals and preferences of a broad group of stakeholders. What is needed to inform decisions are transparent, repeatable, and accessible tools and open-source data for resource-poor nations to identify where ecosystems matter most for people (United Nations Office for Disaster Risk Reduction [UNDRR], 2019).

Here we present the results from a coastal hazard and social vulnerability analysis for The Bahamas using the InVEST Coastal Vulnerability model (Sharp et al., 2018), and we discuss how the results from this analysis were used to inform several planning efforts to build coastal resilience in the country. Modeling was focused on addressing three fundamental questions that decision-makers often consider when implementing nature-based coastal protection: (1) where are people at risk from coastal hazards in The Bahamas? (2) how might sea-level rise (SLR) change the distribution of risk across the country? and (3) where are coastal and marine ecosystems providing protection currently, and under future SLR for the most socially vulnerable populations? In the subsequent sections we first detail the modeling methodology and data collection. Next we report on risk results at the national and island scales, highlighting the drivers of risk, the role of ecosystems in reducing risk, and implications of sea-level rise. We then explore how these results were used to inform post-disaster decision-making for Hurricanes Matthew (2016) and Joaquin (2015), and a 2-year sustainable development planning initiative for Andros Island led by the Office of the Prime Minister. Our results illustrate the diversity and versatility of information that can be provided by a relatively simple approach and tools that integrate multiple components of vulnerability and resilience for coastal communities.

MATERIALS AND METHODS

We explored risk to coastal communities in The Bahamas by combining results from a hazard analysis that considered the role of ecosystems in reducing impacts from flooding and erosion with demographic information about vulnerable populations. In the sections that follow, we introduce the study area and the theory behind the InVEST Coastal Vulnerability model. We then describe how data were collected, how model variables were parameterized, and we introduce scenarios that were tested in this work. Finally, we discuss how social variables were mapped and related to the hazard index to understand risk to people and benefits of coastal ecosystems.

Study Area

Bahamians are highly dependent on the services provided by the country's extensive marine and coastal ecosystems which include (but are not limited to) coral reefs, mangrove forests, coastal coppice forests (dry broadleaf evergreen forests), Caribbean pine forests and seagrass beds (**Figure 1A**). More than three-quarters of the country's GDP comes from the tourism sector, and an estimated quarter of Bahamian households derive some income from fisheries (Moultrie et al., 2016; CIA World Factbook, 2017; Environmental Resources Management Inc [ERM], 2017). While not reflected directly in GDP or income, storm protection from coastal habitats is a critical ecosystem service in The Bahamas, given the country's low terrain elevation and location in a hurricane-prone area. An estimated 15% of the country is within a 3 m SLR flood zone (based on SRTM 1 Arc Second Global elevation data), which is the highest flood risk of any country within the insular Caribbean (The Nature Conservancy, unpublished). Risk is compounded by the fact that the majority of the population lives in low elevation coastal zones less than 10 m above sea level, within 5 km of the coastline (Center for International Earth Science Information Network [CIESIN], and Columbia University, 2013; United Nations Economic Commission for Latin America and the Caribbean [UNECLAC], 2014). Furthermore, the Bahamian archipelago is vast, with 700 islands and over 2,400 cays. Nineteen islands support 90% of the population, concentrated on three main islands in the Northwestern part of the country (New Providence, Grand Bahama, and Abaco). The remainder of the population is distributed on three less populated islands (Eleuthera, Andros, and Exuma) and thirteen sparsely populated islands and cays, primarily in the Southeastern part of the country (Bahamas Department of Statistics, 2017; **Figure 1B**). Many of the sparsely

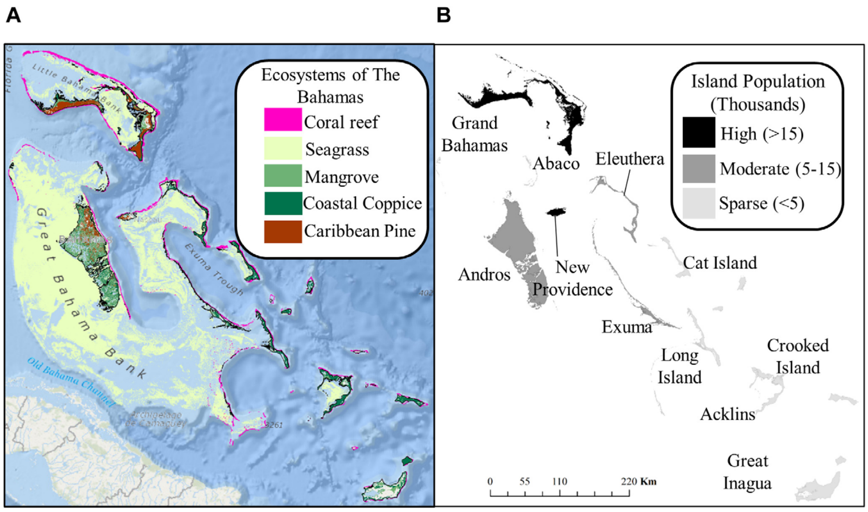


FIGURE 1 | (A) Major coastal, nearshore, and marine ecosystems of The Bahamas. **(B)** Islands of The Bahamas and distribution of the population. The attribution for the basemap is ESRI (2012).

inhabited islands are remote, and difficult to access for disaster management, emphasizing the importance of storm damage mitigation by habitats in reducing costly disaster aid.

Modeling Coastal Hazard

To estimate risk from coastal hazards to people throughout The Bahamas now and with future SLR, we used the InVEST Coastal Vulnerability model. The Coastal Vulnerability model is a decision support tool that uses an index-based approach to understand the relative risk of communities to coastal hazards and identifies where habitats have the greatest potential for providing coastal protection (Arkema et al., 2013; Langridge et al., 2014; Hopper et al., 2016; Cabral et al., 2017; Sharp et al., 2018). The model builds on previous, similar indices that account for biophysical and climatic components governing exposure to flooding and inundation from coastal hazards (e.g., Gornitz, 1990; Cooper and McLaughlin, 1998; Hammar-Klose and Thieler, 2001), by explicitly considering the role of ecosystems in providing coastal protection and incorporating information about people, property and other relevant metrics in the framing of risk.

We assessed risk from coastal hazards to coastal communities at a 250 m^2 spatial resolution along the entire coast of The Bahamas for several SLR and habitat scenarios. We used the InVEST Coastal Vulnerability model to compute a hazard index that ranked the relative exposure of the shoreline to flooding and erosion based on the following variables: habitat type and extent, coastal elevation, wave exposure, shoreline type, storm surge potential, and SLR (see below for a more complete description of each variable). For each 250 m coastal segment, the variables listed above were assigned ranks from lowest exposure (rank = 1), to highest exposure (rank = 5) based on a combination of absolute and relative rankings of modeled and observed data (**Table 1**). The final coastal hazard index was calculated by taking the geometric mean of the ranked variables (where R = rank, and all variables given equal weighting). The results are the relative exposure to flood and erosion hazards for each 250 m coastal segment compared with all other segments countrywide, across all scenarios for SLR and habitat (see below).

$$\text{Hazard Index} = (R_{\text{Habitats}} R_{\text{ShorlineType}} R_{\text{Relief}} R_{\text{SLR}} R_{\text{Waves}} R_{\text{SurgePotential}})^{1/6}$$

Following Hammar-Klose and Thieler (2001), we used a multiplicative model for the exposure index instead of an additive one because coastal processes and interactions among components of coastal ecosystems are inherently non-linear (Barbier et al., 2008; Koch et al., 2009), and because a linear formulation is susceptible to "eclipsing," where one variable can be low but the overall index is not (Ott, 1978; Swamee and Tyagi, 2000). The geometric mean is used as the aggregation function in a variety of environmental index models and produces intuitive results because it resolves to the same scale as the inputs. For example, Landwehr and Deininger (1976) demonstrate its favorable predictive ability over other formulations in the context of water quality. Arkema et al. (2013) find strong correlation between hazard index values derived using this multiplicative formula and observed data on hazard events and losses for the coastal United States.

To map hazard we classified the full distribution of values from the hazard index for all segments and scenarios (ranging from 1 to 5) into three groups. We demarcated areas of highest hazard (>2.667 = top 50% of the distribution), intermediate hazard (2.316–2.667 = middle 25–50% of the distribution), and lowest hazard (<2.316 = lowest 25% of the distribution). In addition, cutoffs for categorical breakdowns were informed by empirical observation of currently vulnerable areas in The Bahamas based on damage reports following recent hurricanes (e.g., Caribbean Disaster Emergency Management Agency [CDEMA], 2016a,b,c,d; Pacific Disaster Center [PDC], 2016). In subsequent

TABLE 1 | Coastal hazard index variables and ranks.

Rank variable	Very low exposure 1	Low 2	Moderate 3	High 4	Very high exposure 5
Natural habitats	Coral reef, Coppice, Mangrove (tall)		Mangrove (short), Caribbean Pine	Seagrass	
Shoreline type		Seawall	Rock	Mud	Sand
Relief	First quantile	Second quantile	Third quantile	Fourth quantile	Fifth quantile
Wave exposure	First quantile	Second quantile	Third quantile	Fourth quantile	Fifth quantile
Surge potential	First quantile	Second quantile	Third quantile	Fourth quantile	Fifth quantile
Sea-level change	0–40 cm	41–80 cm	81–120 cm	121–160 cm	161–200 cm

Variables may be given absolute (e.g., natural habitats, shoreline type, and sea-level change) or relative ranks (e.g., relief, wave exposure, and surge potential). Ranks for the relief, wave exposure, and surge potential are based on a quantile breakdown (first quantile = 0–20% percentile, etc.) calculated from the full distribution of values for all 250 m shoreline segments across The Bahamas. Ranks for natural habitats, shoreline type, and sea-level change are assigned by the user (see "Materials and Methods" for more detail) This was adapted for The Bahamas from Supplementary Table 1 in Arkema et al., 2013, and Table 4.1 in Sharp et al., 2018 and applied using the InVEST Coastal Vulnerability model.

sections we use the terms highest, intermediate and lowest to express relative exposure to coastal hazards. We then combine the exposure results from the coastal hazard index with demographic data we used to map vulnerable populations to estimate risk from coastal hazards for coastal communities throughout The Bahamas. This is described below in subsequent sections on mapping and quantifying risk to coastal communities.

Data Collection for Hazard Modeling

Model inputs were compiled from globally available, countrywide, and island-level datasets and included variables for coastal and nearshore habitats, relief, wave exposure, shoreline type, and surge potential.

Habitat

We identified five main types of coastal and nearshore habitats that occur along the coast of The Bahamas that may provide some degree of coastal protection: coral reefs, seagrass beds, mangrove forests, coastal coppice forests, and Caribbean pine forests. The hazard index ranks habitats based on differences in their morphology and expected ability to provide protection from erosion and flooding by dissipating wave energy, attenuating storm surge, or anchoring sediments, for example. In addition, the index accounts for greater protection provided by co-occurring habitats (Guannel et al., 2016) and assigns a distance over which different types of habitats will provide protection for coastlines (i.e., "protective distance") (Arkema et al., 2013; Sharp et al., 2018). The coastal vulnerability model also requires spatial information (shapefiles) about the type and extent of habitats. In The Bahamas, we created composite habitat maps from multiple sources, years, and spatial resolutions and extents in order to reconcile incongruences across input layers (**Table 2**).

To map the distribution of coral reefs we used three sources: (1) the National Coral Reef Institute (NCRI) at Nova Southeastern University Oceanographic Center dataset which covers the barrier reef off the East Coast of Andros Island (The Nature Conservancy [TNC], and The National Coral Reef Institute [NCRI], 2010), (2) the Millennium Coral Reef Mapping Project data covering the rest of The Bahamas (Andréfouët et al., 2006; UNEP-WCMC et al., 2010), and (3) data from the Marine Spatial Ecology Lab at the University of Queensland, Australia (Marine Spatial Ecology Lab [MSEL], 2005). Reef were also filtered by depth such that reef deeper than 20 m were excluded from the analysis. For the expected wave heights, a depth of 20 m was considered as the threshold beyond which the wave-bed interaction was negligible. As a result, it was primarily the reef crest that was included in the analysis, which is thought to provide the majority of reef related coastal protection services (Ferrario et al., 2014). Coral roughness was assumed to be the same across the study area, which did not account for changes in rugosity due to coral composition or degradation.

Seagrass coverage was compiled from different datasets, one for the area around Andros Island (The Nature Conservancy [TNC], and The National Coral Reef Institute [NCRI], 2010) and two for the remainder of the county (The Nature Conservancy [TNC], and The University of South Florida, 2007; Knowles et al., 2017).

To map the distribution of coastal vegetation including mangrove forests, coastal coppice forests and Caribbean Pine forests, we used RapidEye (2009, 5 m) and Landsat 5 and 7 (2005, 30 m) satellite imagery classified by TNC for Andros Island. We also used forest cover digitized from 1969 British Admiralty Lands and Survey Department topographic maps for the remainder of the country. These older maps were manually updated to capture changes in forest cover in the past 50 years discernable from satellite imagery. Mangrove forests were then divided into two categories: (1) tall mangrove and (2) mangrove swash/swamp (characterized by lower canopy heights and more open mudflat between plants). Thus, four types of coastal vegetation were input into the model (**Table 1**).

All habitat data were converted to Esri shapefiles with linear units in meters (WGS 1984 UTM Zone 18N projection). As described above, country-wide habitat maps combined the best available data, which varied in age and resolution (Moss and Moultrie, 2014; Knowles et al., 2017). Attempts made to update older datasets are described above, and the model can be reapplied as additional and more accurate data become available.

TABLE 2 | Data sources for Coastal Vulnerability model inputs.

Model input		Year	Extent	Resolution	Source
Natural habitats	Coral Reef	2010	Andros Island	5 m	The Nature Conservancy and National Coral Reef Institute at Nova Southeastern University dataset
		2010	Global	30 m	Millennium Coral Reef Mapping Project (Landsat 7)
		2005	The Bahamas	30 m	Landsat imagery classified by the Marine Spatial Ecology Lab at the University of Queensland, Australia
	Seagrass	2010	Andros Island	5 m	The Nature Conservancy and National Coral Reef Institute at Nova Southeastern University dataset
		2007	The Bahamas	30 m	Landsat 7 imagery classified by The Nature Conservancy and the University of South Florida
	Mangrove, Coppice, Caribbean Pine	2009	Andros Island	5 m	Rapideye imagery classified by The Nature Conservancy
		2005	Andros Island	30 m	Landsat 5 and 7 imagery classified by The Nature Conservancy
		1969	The Bahamas	1:50,000	British Admiralty Lands and Survey Department topographic maps
Relief	Digital elevation model (30 m)	2014	Global	30 m	Shuttle Radar Topography Mission
		2004	Global	1 km	World Resource Institute
		Variable	The Bahamas	Variable	Nautical charts: National Oceanographic and Atmospheric Administration, NAVCHARTS, etc.
Wave exposure		2005–2010	Global	50 km	National Oceanographic and Atmospheric Administration WaveWatch III
Shoreline type	Coastal geomorphology	1970, 2000, varies	The Bahamas	Vector	Digitized from the Department of Lands and Surveys Topographic Maps (1970) and Landsat 7 (2000) by The Nature Conservancy and digitized from aerial imagery (various years) by The Natural Capital Project
Surge potential	Continental shelf	2005	Global	Vector	Continental Margins Ecosystem (COMARGE) effort in conjunction with the Census of Marine Life

Habitat Ranks and Protective Distances

Each habitat type was assigned a rank based on differences in morphology and expected ability to prevent erosion and attenuate waves and storm surge. A rank of "1" offers the greatest protection, "4" the least, and "5" designates no protection afforded by habitat. We did not include any process-based reduction or attenuation function of waves or surge in the habitat ranking system. Habitat ranks are presented in **Table 1** and are based on expert judgment and the peer-reviewed literature (e.g., reviewed in Shepard et al., 2011; Arkema et al., 2013; Spalding et al., 2014; Narayan et al., 2016). A habitat-specific "protective distance" was also defined to indicate the extent of coastline likely receiving protection from a given habitat type. These distances are essentially a technical shortcut, rather than an ecological or hydrodynamic parameter. They allow us to designate which coastline segments are protected by patches of habitats located at different distances from the grid cells, given that the model does not take into account the numerous factors (depth, channel configuration, distance from the coast, etc.) that could influence the distance over which effects of these habitats may be prominent (Arkema et al., 2013; Sharp et al., 2018).

Lastly, we included in the index the protection provided to coastal segments by more than one habitat type (Guannel et al., 2016). For example, some shorelines may have only coral reefs, while other areas are fringed by mangroves and seagrass, as well as corals. The ranks have been assigned in such a way that multiple co-occurring high-ranking habitats (e.g., seagrass and short mangrove) perform better than either one alone, but do not perform as well as a lone low-ranking habitat such as coral reef (Sharp et al., 2018). Our ranking approach is a first attempt to incorporate the role of multiple habitats in reducing coastal vulnerability over such a large geographic scale and is flexible enough to be refined as future research in this field emerges.

Relief

Coastal elevation (i.e., relief) is an important indicator for potential inundation during storm events. To calculate a relief rank, we created a seamless topo-bathy Digital Elevation Model (DEM) from three datasets of varying spatial and temporal resolutions: (1) 30 m globally available SRTM version 3 topography data (NASA LP DAAC, 2014), (2) 1 km bathymetry for The Bahamas from the World Resource Institute (World Resources Institute [WRI], 2004), and (3) digitized nautical chart soundings at different scales from across The Bahamas (NOAA, NAVCHARTS, etc.). For each shoreline segment, elevation (i.e., relief) was averaged within a 2,000 m averaging radius and assigned a relative rank of 1–5 based on the full distribution of values (the model uses percentile breaks 20, 40, 60, and 80 to categorize the distribution by default). The neighborhood mean was selected to capture significant changes in elevation along the shoreline without being influenced by inaccuracies in relatively coarse topographic data inputs (DEM) (**Table 2**).

Wave Exposure

Waves are an important factor influencing the erosion and flooding of shorelines. The wave exposure was estimated from the average power of the 10% largest waves (in height) encountered in each of the 16 cardinal directions from a given shoreline segment (Sharp et al., 2018). Wave heights were in turn extracted from a globally available dataset; the National Oceanographic and Atmospheric Administration (NOAA) WAVEWATCH III hindcast re-analysis results for an 8-year period (2005–2013) (Tolman, 2009). Wave exposure is calculated differently by the model for oceanic and locally wind-generated waves, as sheltered coastline segments are exposed only to local waves. The final relative ranks of 1–5 are assigned based on the full distribution of wave power values observed in The Bahamas. For example, those shoreline segments with wave power values falling in the 0–20th percentiles (i.e., first quantile) of values relative to all shoreline segments in the country were assigned a rank of "1," and those with values in the 81–100th percentiles (i.e., fifth quantile) were assigned a rank of "5" (**Table 1**).

Shoreline Type

Shoreline type describes the composition of the shoreline, which influences susceptibility to erosion. Because the model is a relative ranking model, we sought to capture the dominant shoreline types and highlight differences in their relative susceptibility to erosion. Several datasets were combined to produce a country-wide shoreline type layer. Sandy beaches and rocky shores were digitized by The Nature Conservancy from Department of Lands and Survey Topographic maps (1970) and Landsat 7 imagery (2000) (Knowles et al., 2017). Using the map of mangrove distribution described above, we assumed muddy shorelines where mangrove was dominant. Lastly, we used high-resolution satellite imagery (Google Earth, Microsoft Bing) to fill gaps in the country-wide shoreline type map. Three naturally occurring shoreline types were classified for The Bahamas: sandy beaches were given a rank "5" muddy shorelines ("4"), and rocky shorelines ("3"). Seawalls were given a rank of "2." Comprehensive data on the location of seawalls were not available country-wide. Major seawalls detectable via satellite imagery were included in the model but many smaller seawalls were not. Furthermore, seawalls often have an edge effect where erosion is amplified around the edges. This can be captured by the model, but we chose not to reflect this in our analysis due to the fact that our seawall coverage was incomplete.

Storm Surge Potential

To estimate surge potential, we calculated the cross-shore distance between each segment of coastline and the edge of the continental shelf. The distance to the shelf is a proxy for storm surge potential based on the well-known fact that a shallow bathymetry promotes the "piling up" of water during storm events, causing the phenomenon of storm surge (Resio and Westerink, 2008). To calculate this proxy, we used a polygon representing the edge of the continental shelf that was prepared by the Continental Margins Ecosystem (COMARGE) effort in conjunction with the Census of Marine Life. Since storm surge is a critical driver of exposure to coastal hazards in The Bahamas, and the above proxy for surge potential may oversimplify the dynamics of the phenomenon, we also had local experts from the Departments of Meteorology and other relevant agencies review the results obtained via this simple approach. Their experiences with areas significantly affected by previous storms supported our results. Finally, we compared the relative risk of exposure to storm surge, as estimated by the proxy discussed above, to values from the Storm Surge Atlas (Rolle, 1990) available for a portion of the Northern and Central Bahamas. The relative relationship of surge potential across the region agreed qualitatively well between our proxy and the modeling from the Storm Surge Atlas for this region.

Habitat and Sea-Level Rise Scenarios

Habitat Scenarios

A primary goal of this analysis was to quantify the role that natural habitats play in reducing risk to people in The Bahamas. To quantify habitat role, we considered two heuristic scenarios, a "with habitat" scenario that accounts for the protection provided by the current distribution of coastal and nearshore habitats throughout the country, and a "without habitat" scenario where habitat is assumed to be lost, and no longer provide protection. The "without habitat" scenario is intended to evaluate where and to what extent habitats are providing protection to people, and is not intended to represent an actual reflection of the future. To represent a "without habitat" scenario in the model, the habitat rank was changed to "5."

Sea-Level Rise Scenarios

This analysis involved comparing the relative exposure to coastal hazards under current sea levels (2015) against two future SLR scenarios (2040, 2100). We focused primarily on 2040 because this analysis was conducted in the context of a sustainable development planning effort for Andros Island and a national development planning process (Vision 2040) which both had time horizons of 25 years. No local tide gauge data were available to produce spatially explicit rates of SLR within the country. Instead, we looked at the relative change between current and future scenarios assuming uniform rates of SLR across the entirety of The Bahamas. We estimated the relative change in sea-level between timesteps using the projected SLR curve for the highest RCP scenario (2 m rise by 2100) depicted in Figure ES 1 of Parris et al. (2012). To do this we divided the net rise (cm) from the start of the curve (1992) to the end (2100) into quantiles as follows: 0–40 cm rise corresponded to a rank of "1," 41–80 cm "2," 81–120 cm "3," 121–160 cm "4," and 161–200 cm a rank of "5." Using the curve, we estimated the net rise at the current timestep (2015) within the first quantile (~10 cm) and assigned a rank of "1." The projected rise for 2040 (our planning horizon) was ~40 cm and was assigned a rank of "2" (**Table 1**). This is a simple approach to reflect the increased exposure to coastal hazards anticipated as sea-levels rise.

While we were primarily concerned with understanding changes in risk associated with the near-term SLR scenario (2040) which aligned with the planning process, we also considered a longer-term scenario (2100). According to the same approach we used for the current and 2040 scenarios, the projected net rise for 2100 was 2.0 m (Parris et al., 2012) and was assigned a rank of "5." This represents a more extreme increase in sea-levels expected over longer timelines (i.e., 2100) to further investigate the influence of SLR as driver of risk.

Mapping Coastal Communities

To relate hazard to people at risk, first we mapped demographic variables for the country including total population (people/500 m^2), elderly people (>65 years old) and young people (<15 years of age) (per 500 m^2). We classified young people as <15 years of age, which is older than the cutoff of <5 generally cited in the social vulnerability literature. We used the higher age cutoff to capture demographic patterns even in more sparsely populated parts of the country. Demographic data collected during the 2010 Bahamian Census at the supervisory district level, obtained from the Department of Statistics, was distributed spatially using the globally available Nighttime Lights Time Series (National Oceanic and Atmospheric Administration [NOAA], 2011). Light intensity (a proxy for population density) was extracted from all supervisory districts in The Bahamas from NOAA's Nighttime Lights Times Series. For each supervisory district a "demographic metric per unit of light intensity (DM/LI)" was calculated by dividing the demographic variables for that supervisory district by the summed light intensity for that district. The DM/LI ratio was then multiplied by the entire nighttime lights raster for that supervisory district to distribute the demographic variable across the district. This methodology allows the demographic variables to be mapped at a finer resolution by using the relative weighting of the light intensity (Nicholls et al., 2008; Henderson et al., 2011).

Quantifying Risk to People

To assess the vulnerability of people (total population, elderly and young people) in The Bahamas to coastal hazards, we analyzed the overlap between the coastal segments with the highest exposure to coastal hazards and the metrics described above. To do this we estimated the total number of people, number of elderly and young individuals living within 1 km of the coast for each 250 m^2 shoreline segment from the coastal hazard index (after Arkema et al., 2013). No estimates of expected population growth were used in this study, so the values represent current (2010) population only.

RESULTS

In the sections that follow we start with a review of key findings at the national-scale and then focus more in depth on island-scale results. In particular we report on the spatial distribution of risk, the drivers of risk, and the potential for coastal and nearshore ecosystems to provide protection to people now and with future SLR. In the discussion we describe how these results were used in a number of different decision-making processes in The Bahamas.

National-Scale

Spatial Distribution of Risk

Modeled results indicate that nearly one fifth of the coastline and nearly two in ten Bahamians are currently at highest risk of exposure to coastal hazards (**Figures 2A,B**). With modeled SLR, we found that the extent of shoreline most exposed to coastal hazards would more than double, and the total population would nearly triple (with more than 10% of the population, >40,000 people, living in highest risk areas) (**Figure 2B**). Modeled estimates of population assume a constant 2010 population as no projections were available for 2040, but given that population is increasing in The Bahamas these results likely underestimate future risk.

Coastal Protection Provided by Ecosystems

Coastal and nearshore ecosystems occur along almost the entire coastline of The Bahamas, often with multiple habitats fronting sections of shoreline (e.g., coral reef backed by seagrass and mangrove) (**Figure 1A**). Our results suggest that if these habitats are lost, even under current sea-levels, the length of shoreline highly exposed to hazard throughout the country would quadruple (**Figure 2B**). With habitat loss and modeled SLR, the length of shoreline at highest exposure increases fivefold (**Figure 2B**), putting an estimated quarter of the population at highest risk. These results highlight the important role ecosystems may be playing in providing coastal protection now and in the future.

Island-Scale

Distribution and Drivers of Coastal Hazard

We found the most exposed coastline in The Bahamas located predominantly along the side of islands that sit on extensive shallow banks, where the potential for significant storm surge is high (**Figures 2A**, **3**). Notable examples included the north coast of Grand Bahama and west coast of Abaco, which sit on the Little Bahamas Bank, the west coast of Andros Island, located on the Great Bahama Bank, and the western coast of Acklins and Crooked Islands, situated on a large shallow lagoon called the Bight of Acklins (**Figure 2A**). Modeled results indicate that, relative to the rest of the country, these four islands have the greatest proportion of highest hazard shoreline (**Figures 4**, **5**). New Providence Island, the most densely populated in the country and the seat of the capitol city Nassau, is the next most exposed island in The Bahamas, with nearly a tenth of its total shoreline currently highly exposed. As with the previous examples, the most exposed areas on New Providence are primarily along the southern coast of the island where it is positioned on a shallow tongue of the Great Bahama Bank. Our modeling suggests that, in addition to surge potential, low elevations and soft, erodible sediments are key factors driving risk on islands with large proportions of exposed shoreline (**Figure 3**).

Shoreline of other islands, such as Great and Little Inagua, Mayaguana, San Salvador, and the Ragged Island chain, are relatively less exposed, compared to the rest of the country (**Figures 2A**, **4**, **5**). On these islands, we found that exposure is mitigated by relatively higher elevations, lower potential for

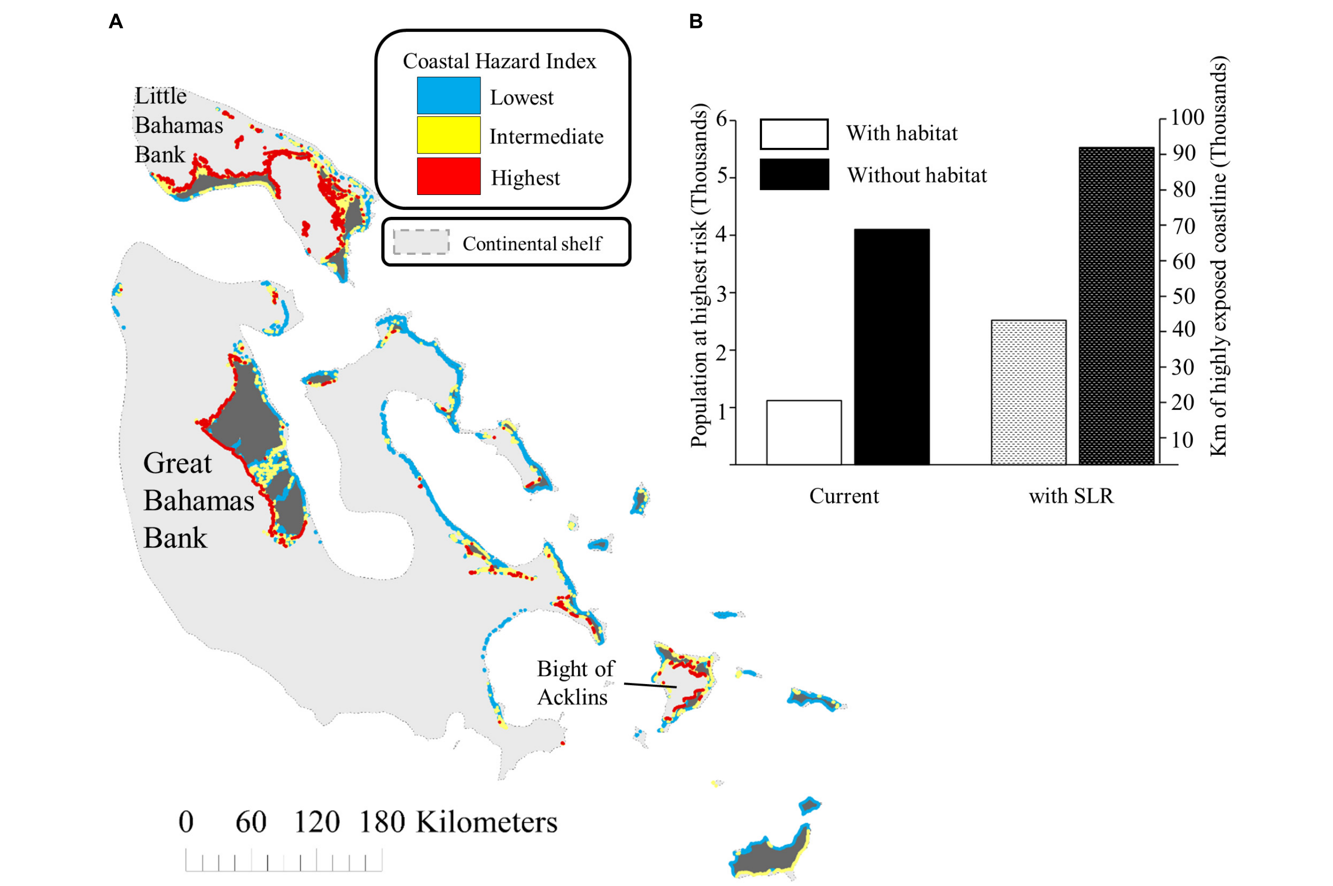


FIGURE 2 | (A) Relative exposure to coastal hazards for The Bahamas. Storm surge is a key driver of exposure for The Bahamas and this is reflected in the modeled results. The greater the distance from land to the edge of the continental shelf (in gray), the greater the potential for exposure to storm surge. **(B)** Length of highly exposed coastline and number of people at the highest risk with and without coastal and nearshore habitats, currently and with future SLR. Results are represented using the same set of bars for both metrics because at the national scale these variables are highly correlated.

exposure to storm surge, and in some cases rocky shorelines less prone to erosion. In addition, the presence of coastal and nearshore habitats is vital for protecting these islands. For example, our results indicate that the relatively low exposure of San Salvador and Great Inagua Islands is attributable in large part to attenuation of waves by coastal habitats. If these habitats were lost, nearly half of the coastline of these islands would be classified as highest exposure areas (**Figure 4**).

Coastal Protection Provided by Ecosystems

Modeled results showed that coastal and marine ecosystems are crucially important for reducing exposure to coastal hazards for every island in The Bahamas. We found that ecosystems provide coastal protection for islands where exposure is inherently high due to other factors (elevation, storm surge potential, etc.), and are equally important for maintaining low exposure of other islands. For example, Grand Bahama Island has the greatest extent of highly exposed shoreline of any island in The Bahamas (almost half of the island is at highest risk). However, Grand Bahama also benefits from coastal protection along >300 km of the island's coastline by extensive seagrass beds, coral reef, mangrove, and coastal coppice forests (**Figures 4**, **6**). Our results suggest that if these habitats are lost, almost the entirety of Grand Bahama would be highly exposed relative to the rest of the country (**Figure 4**). In contrast, the shorelines of Great Inagua Island are less exposed to coastal hazard, due in part to higher elevations and lower storm surge potential relative to the rest of the country. But ecosystems, including a fringing reef encircling the island and mangrove forests (**Figure 1A**), are also critical in protecting the island. We found that, like Grand Bahama, the loss of habitats would result in a 50% increase in exposure for Great Inagua Island (**Figure 4**).

Future Scenario (SLR)

This analysis also highlights the potential for ecosystems to play a role in mitigating increased exposure to coastal hazard due to SLR across The Bahamas. While modeled SLR indicates an increase in exposure to every island in The Bahamas, our results suggested that the presence of marine and coastal ecosystems can reduce the extent of highest exposed shoreline significantly for all islands, by up to two-thirds, island dependent (**Figure 4**). For example, the proportion of highly exposed coastline along

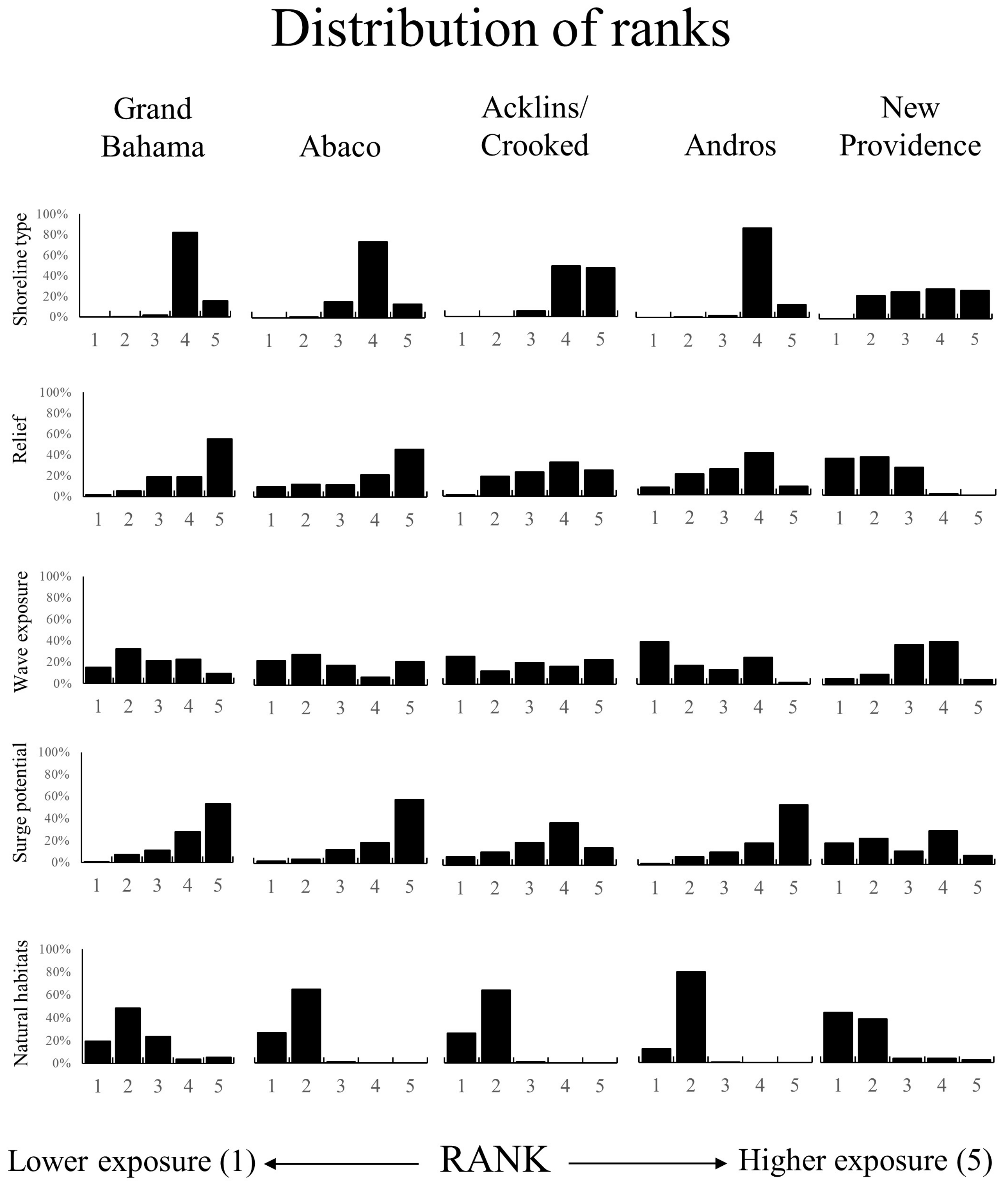


FIGURE 3 | Distribution of ranks for individual variables in the coastal hazard index for each 250 m coastal segment for the islands in The Bahamas with the greatest proportion of highly exposed coastline. A rank of "1" contributes to lowest exposure, while a rank of "5" contributes to highest exposure. These distributions of ranks for individual variables in the coastal hazard index reveal key drivers of exposure across the region. For example, wave exposure ranks are distributed relatively uniformly (from 1 to 5) across each island indicating that this variable may be less critical in influencing differences in exposure among these five islands. However, the shoreline type and surge potential variables are skewed toward higher ranks, indicating that these are important variables driving the relatively high exposure on these islands.

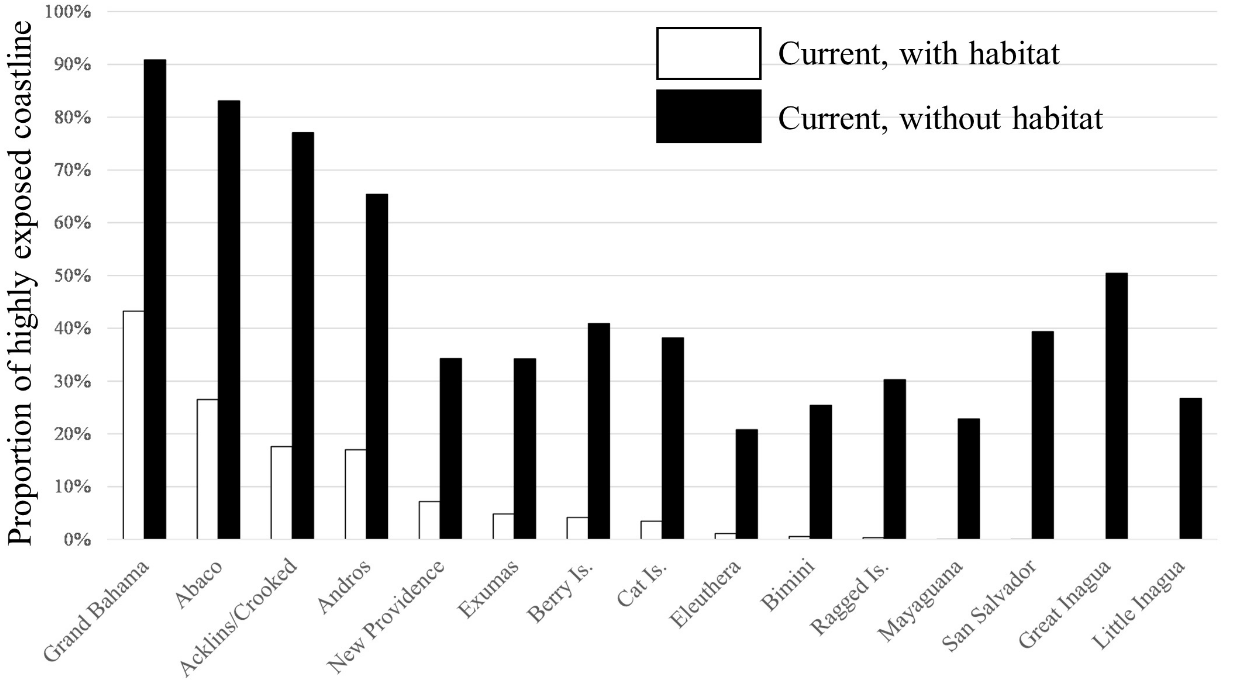


FIGURE 4 | Percentage of highly exposed coastline for each major island in The Bahamas, with and without coastal and marine ecosystems. This is for the current scenario, not accounting for SLR.

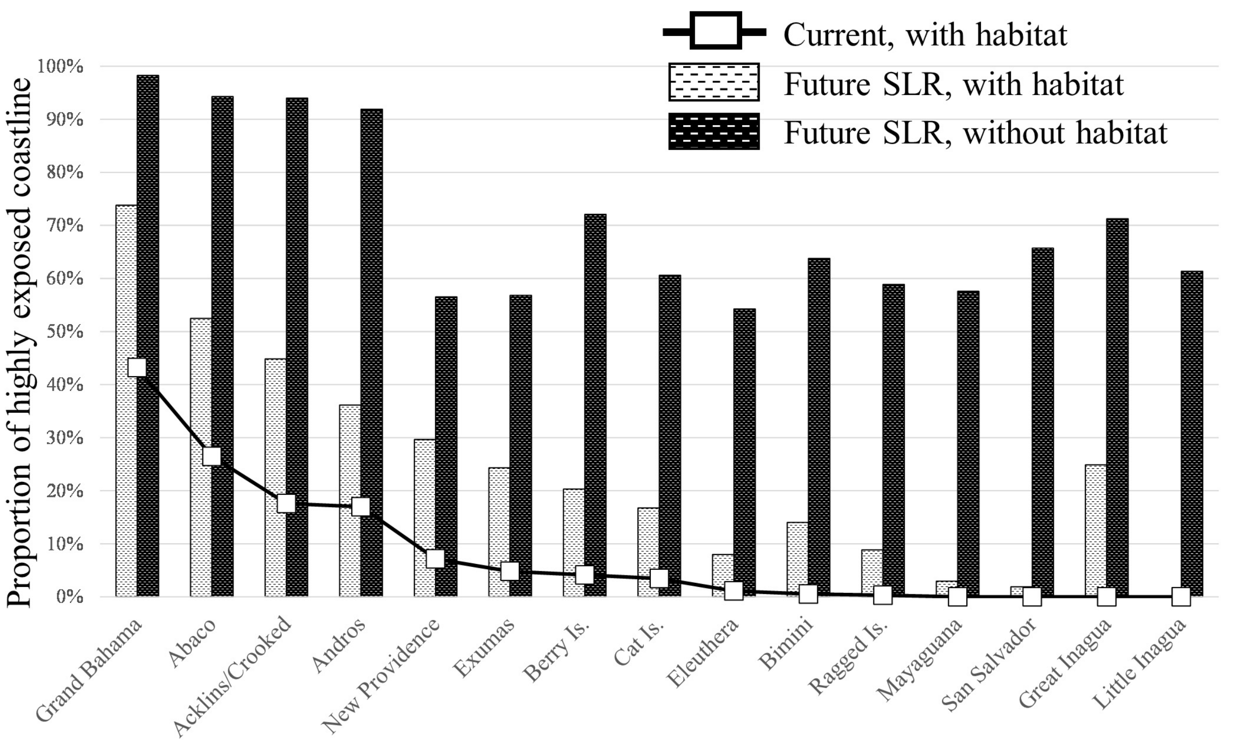


FIGURE 5 | Percentage of highly exposed coastline for each major island in The Bahamas, with and without coastal and marine ecosystems, currently and with future SLR.

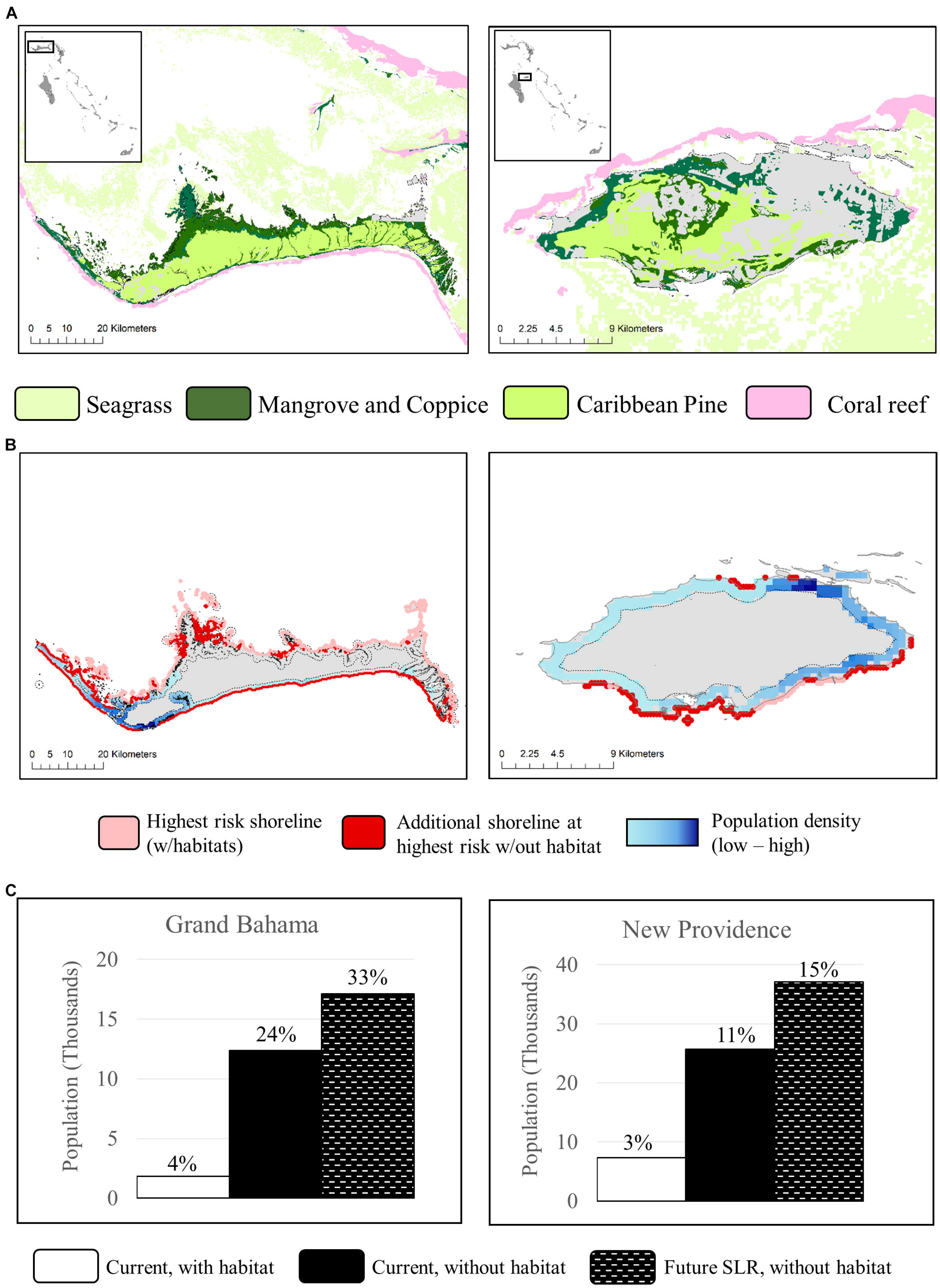


FIGURE 6 | Coastal and marine ecosystems on Grand Bahama and New Providence Islands **(A)** provide protection for people who, if those habitats were lost, would be living along the highest risk shoreline **(B)**. Population within a 1km inland coastal hazard zone is indicated with a dashed line. Bar charts in **(C)** show the total number of people and the percentage of the total island population at highest risk in each scenario (with habitat and current sea-levels, without habitat and current sea-levels, and without habitat and SLR).

Grand Bahama increases by about a third in the SLR scenario, and if compounded by habitat loss, almost the entire island is highly exposed. In contrast, if habitats are kept intact, only three-quarters of the shoreline is at highly exposed under future conditions (**Figure 5**). The difference in exposure with and without habitats suggests that the presence of habitats can reduce the potential increases in exposure to hazards associated with SLR on the island of Grand Bahama by 20% or more (**Figure 5**). While these estimates are relative and reflect modeling assumptions and simplifications, they hold true for all islands in the country (to varying degrees), and thus suggest a crucial role for habitats in mitigating future hazards.

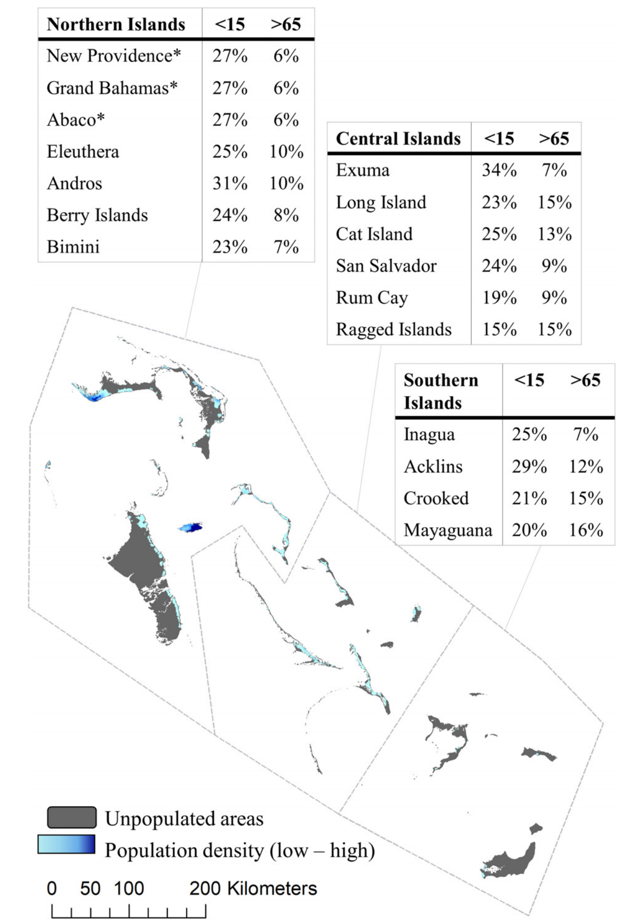
When a more extreme SLR scenario was considered (e.g., year 2100), we found that the qualitative outcomes remained the same: risk increases with SLR and loss of habitat, but the magnitude of risk increases substantially more with greater SLR. We also found that under a more extreme SLR scenario, the potential for habitats to provide protection is reduced. Country-wide, our results suggests that coastal protection provided by habitats reduces the extent of shoreline (and number of people) at high risk by over half under the moderate SLR scenario (2040) (**Figure 2B**), and by a third in the more extreme SLR scenario (2100).

Coastal Protection Provided by Ecosystems to People

We found that coastal ecosystems currently provide protection for nearly fifteen percent of Bahamians who, if these habitats are lost, will be at the highest risk of exposure to coastal hazards. For all the main inhabited islands of The Bahamas, the fraction of the population at highest risk doubles, and in many cases triples or increases by an order of magnitude if coastal habitats are lost.

Modeled results suggest that on some islands large numbers of people benefit from the coastal protection provided by ecosystems. For example, only an estimated 4% of the population of Grand Bahama Island is currently living in highest risk areas, but if habitats are lost near a quarter of the island's population will be at highest risk (**Figure 6**). Similarly, on Exuma Island only an estimated 2% of the population is currently living in highest risk areas, but this increases to over one third if habitats are lost. On other islands, the loss of habitats results in dramatic increases in risk for an already disproportionately high risk population. For example, on Abaco Island a quarter of the population currently lives in highest risk areas, but if ecosystems are lost, nearly the entire island will be at highest risk. On New Providence Island, habitats provide protection to some of the most densely populated shoreline in The Bahamas. Our results indicate that roughly a tenth of the island's population is living in an area that may become highest risk if existing habitats are lost. This corresponds to an estimated 18,000 people on New Providence alone that are benefiting from risk reduction provided by coastal and marine habitats (**Figure 6**).

Furthermore, our analysis highlights where ecosystems are providing protection to socially vulnerable populations. Nationally, roughly one third of the 2010 population in The Bahamas is under 15 years of age, and almost a tenth is elderly (>65). The three most populated islands in the country (New Providence, Grand Bahama, and Abaco) reflect the national average (**Figure 7**). However, the fraction of elderly is double (or more) on many other islands, and generally increases in the more remote and sparsely populated southern and central islands. Acklins/Crooked and Andros Islands have the highest proportions of both elderly and young people, relative to the other islands in the country (**Figure 7**). These islands also have large extents of highest risk shoreline. And the proportion of high risk shoreline increases significantly with habitat loss (**Figures 4**, **5**, 7).



Northern Islands	<15	>65
New Providence*	27%	6%
Grand Bahamas*	27%	6%
Abaco*	27%	6%
Eleuthera	25%	10%
Andros	31%	10%
Berry Islands	24%	8%
Bimini	23%	7%

Central Islands	<15	>65
Exuma	34%	7%
Long Island	23%	15%
Cat Island	25%	13%
San Salvador	24%	9%
Rum Cay	19%	9%
Ragged Islands	15%	15%

Southern Islands	<15	>65
Inagua	25%	7%
Acklins	29%	12%
Crooked	21%	15%
Mayaguana	20%	16%

FIGURE 7 | Total population for The Bahamas, as well as elderly and young populations, were mapped using the Nighttime Lights Series (NOAA) for the 2010 population census.

DISCUSSION

The number of studies exploring the coastal protection benefits of ecosystems has grown tremendously in recent years (e.g., Arkema et al., 2013; Ferrario et al., 2014; Spalding et al., 2014; Narayan et al., 2016; Beck et al., 2018). There are, however, few examples of where risk reduction provided by habitats is linked with the socially vulnerable communities that stand to benefit the most, and even fewer examples of where such socio-ecological science has led to on-the-ground investments in conservation or restoration of ecosystems for risk reduction. In a world with rising sea-levels, growing coastal populations, and coastal development threatening coastal ecosystems, innovative approaches are needed that center nature-based protection in a broader socio-ecological framework and provide accessible and transparent tools for decision-makers to explore alternatives. Here we present a modeling approach that allows for assessment of where coastal ecosystems matter most for people now and under future SLR scenarios, and apply it on a timeline to inform coastal management decisions. Modeled results suggest that the magnitude of risk reduction provided by habitats now and under future SLR is substantial. Additionally, our results highlight specific places where such ecosystem-based risk reduction is especially important. Thus, maintaining existing coastal and marine habitat distribution and avoiding future habitat degradation could be among the most effective public policy decisions, both for disaster risk management and climate change adaptation.

The significant contribution ecosystems can have to risk reduction has been shown in previous studies (Alongi, 2008; Barbier et al., 2008; Costanza et al., 2008; Zhang et al., 2012; Arkema et al., 2013; Ferrario et al., 2014; Spalding et al., 2014; Narayan et al., 2016; Beck et al., 2018; Reguero et al., 2018b). Similarly, in this study, we estimated that coastal habitats reduce, by more than half, the extent of shoreline and number of people at highest risk to coastal hazards in The Bahamas under a future SLR scenario. However, the ability of habitats to provide coastal protection depends on their morphological characteristics, distribution, and condition, as well as the forcing conditions they are subject to, all of which make it difficult to generalize where and when natural features will protect people and property (Costanza et al., 2008; Ruckelshaus et al., 2016; Arkema K. K. et al., 2017). We advance the findings from previous studies with methods and approaches to map spatial variation in risk reduction in order to inform spatial planning and decision-making. We identify low-lying locations with highly erodible substrates and multiple habitats as being especially important places to prioritize for conservation in order to maintain coastal protection benefits into the future, especially with sea-level rise. By highlighting the conditions under which ecosystems may be most crucial for coastal resilience, our study can help support more widespread understanding and implementation of nature-based infrastructure in The Bahamas and other countries.

While understanding the physical and ecological factors that enable ecosystems to reduce impacts from coastal hazards is important, a central part of quantifying coastal protection provided by habitats is determining where people stand to benefit the most. Frequently the coastal protection literature tends to focus on the biophysical factors that contribute to risk reduction provided by ecosystems rather than the societal benefits (Arkema K. K. et al., 2017). The results of this study address this gap by synthesizing hazard models, climate scenarios, demographic information and ecological data. Importantly, we identified those locations where the most socially vulnerable populations co-occur with ecosystems that reduce risk, such as elderly populations on remote islands. Age (both elderly and young) is one of the strongest indicators of social vulnerability to coastal hazards both in terms of mortality during storm events and post-disaster recovery, a metric which holds true for resourced and poor nations alike (Cutter et al., 2003; Boruff et al., 2005; Peacock et al., 2012; Arkema K. K. et al., 2017). For example, nearly 60% of the fatalities that occurred in Louisiana during Hurricane Katrina were among the elderly. And mortality increased with age during Hurricane Sandy, with over 30% of deaths occurring in people >65 years of age (Jonkman et al., 2009; Diakakis et al., 2015).

In addition, the population of elderly people tends to be higher on some remote islands in The Bahamas, as working age people often move to the capital city for access to more jobs. Nature-based solutions provide valuable co-benefits such as food (e.g., fisheries) that may be critical relief in disaster contexts where communities are entirely dependent on local resources for subsistence. Building this type of self-sufficiency is a pillar of disaster risk reduction and sustainable development for island nations like The Bahamas where many vulnerable communities remain beyond the reach of rapid assistance after a disaster (Shultz et al., 2016, 2018). Partly in response to the devastation associated with recent hurricane seasons, the need to build coastal resilience is increasingly being recognized as a national priority for the security of Bahamian communities. As a result, building capacity and providing guidance around when and where to implement sustainable coastal protection is an important focus of ongoing island-scale and national planning efforts (Arkema and Ruckelshaus, 2017; Arkema K. et al., 2017; Lemay et al., 2017; Office of the Prime Minister of The Bahamas [OPM], 2017).

Following Hurricanes Joaquin (2015) and Matthew (2016), which caused hundreds of millions of dollars in damages across The Bahamas, decision-makers from a variety of agencies became interested in how our national maps of coastal risk and ecosystems could help inform their post-disaster reconstruction and resiliency building efforts. Training in spatial planning and integrated management approaches were also identified as priorities (Caribbean Coastal Services LTD [CCS], and SEV Consulting Group, 2016). In response, we developed an online interface to easily share the results from this study and to train non-scientists in the modeling approach and tools[2]. We then used these materials in a week-long workshop to engage Bahamians from diverse sectors, including emergency management, the department of works, and tourism and statistics, among others. Representatives from the Ministry of Works found that the risk assessment could help inform placement of roads, water mains, electrical lines, and other infrastructure in a way that would minimize future damage potential. The National Emergency Management Agency (NEMA) found these national-scale results could help inform disaster preparedness efforts in less studied regions of the country (The Tribune, 2016; Environmental Resources Management Inc [ERM], 2017). Our hazard analysis and custom visualization tool provides a national-scale dataset for mapping risk throughout The Bahamas and for supporting the development of comprehensive and effective science-based policies that reduce risk. Building capacity among decision-makers to engage directly with the

[2] http://marineapps.naturalcapitalproject.org/bahamas/

inputs and outputs of models is a crucial step toward trust and ownership of results that has been found to be essential for their uptake (Clark et al., 2011; Ruckelshaus et al., 2013; Clark et al., 2016).

In a parallel effort, the Government of The Bahamas and the Inter-American Development Bank (IDB) used the results from the coastal hazard analysis to help inform the development of a Climate-Resilient Coastal Management and Infrastructure Program. The program, financed by a United States $35 million loan from the IDB, is funding pilot projects on several islands (Lemay et al., 2017), including on the island of Andros where mangrove restoration will be evaluated as a natural coastal protection strategy. At the time of the loan, the authors of this study had been working on Andros for several years to design a sustainable development plan (Arkema and Ruckelshaus, 2017; Office of the Prime Minister of The Bahamas [OPM], 2017). The outcome of the planning process was a roadmap communicating stakeholder desires for investments in infrastructure that would draw on and leverage the island's wealth of natural resources (Office of the Prime Minister of The Bahamas [OPM], 2017), such as the natural-based coastal protection to be financed by the IDB loan. To inform the loan, we used the hazard index as a screening tool to identify high risk, populated areas where mangrove restoration had the potential to provide a cost-effective natural approach to coastal protection (Arkema K. et al., 2017). This was followed by physics-based wave modeling (Roelvink et al., 2010) to quantify the extent of mangrove needed to attenuate waves under different storm conditions and to inform approximate project costs (Arkema K. et al., 2017). Using multiple models to address management questions at different scales, we were able to understand where and under what conditions ecosystems were most appropriate while responding to the short timeline required for the project feasibility assessment. Thus, this study is a promising example for other communities and countries of how application of relatively simple models that link social, ecological, and physical science can be used to inform on-the-ground implementation of investments in ecosystem-based projects for coastal risk reduction.

A third outcome of this study was an opportunity to strengthen the knowledge base of local communities about risks from coastal hazards, climate adaptation, and natural solutions. For example, the settlement of Lowe Sound located at the northern tip of Andros was devastated by the 4.5 m storm surge of Hurricane Matthew. Many years ago when a coastal road was put in, all the mangroves were cleared and a small, approximately 0.6 m seawall was put in to stabilize the shoreline in place of the vegetation. Lowe Sound was identified in the results from our hazard analysis as one of the most exposed locations in the entire country, driven by high storm surge potential, low-lying elevation, and the absence of buffering ecosystems. Residents were shocked to learn that the shallow bank, which they had thought provided protection against storms, actually made them uniquely vulnerable to storm surge. This highlights the importance of educating people living in high risk areas with regard to drivers of coastal hazard. It also underscores the importance of engaging communities in solutions to hazards and climate change (Clark et al., 2011, 2016; Scyphers et al., 2015), such as the IDB loan to finance mangrove restoration. Of course mangrove restoration and conservation, like all nature-based and traditional coastal protection strategies, have their limitations. And as storm size increases, habitats may have less influence on water flow and/or be more likely to be themselves destroyed by the force of waves and surge. Given this, multiple strategies are needed. As climate impacts increase, relocation is anticipated to be part of a comprehensive climate adaptation strategy for many small island developing states (SIDS) (Nansen Initiative, 2015; Mycoo, 2017). However, few nations have policy guidelines to govern the process, and most SIDS currently take an *ad hoc* approach to relocation without considering exposure of the new settlement locations (Mycoo, 2017; Thomas and Benjamin, 2018). Our analysis and approach can help to provide what is missing in many countries – local geospatial data mapping vulnerability for identifying high risk areas (Thomas and Benjamin, 2018) and for engaging communities in developing socially and biophysically feasible solutions.

The transparency of modeled inputs and outputs, and the ability to quickly test different climate and development scenarios make the InVEST Coastal Vulnerability model an effective tool to engage with stakeholders and communicate with scientists and non-scientists alike (Arkema et al., 2013; Langridge et al., 2014; Hopper et al., 2016; Arkema K. K. et al., 2017; Cabral et al., 2017; Office of the Prime Minister of The Bahamas [OPM], 2017). However, there are also several important limitations with the approach. For example, using a proxy for surge potential may oversimplify storm dynamics, especially as they relate to superstorm-supersurge expectations, which could mishandle exposure and vulnerability of certain coastal areas. Furthermore, the dynamics associated with major storms are complex and can result in unexpected scenarios such as the negative surge associated with hurricane Irma (Revesz, 2017). The habitat ranks represent differences in their relative ability to attenuate water flow, but these are based on literature review and ultimately lack information about specific mechanisms and robust empirical validation. The index is most appropriate for understanding relative differences in risk reduction provided by ecosystems along the shoreline and requires assumptions about how far inland exposure to hazards will propagate. We also limited our exploration of climate impacts to sea-level rise alone when in fact many more climate variables, including changes in the intensity and frequency of storms, effects of ocean acidification and warming on reefs, are likely to influence risk to coastal hazards. Lastly, many factors have been identified as key drivers of social vulnerability, such as gender, livelihood, social capital, and wealth (Cutter et al., 2003; Boruff et al., 2005; Peacock et al., 2012; Rhiney, 2015; Ghosal, 2016; Arkema K. K. et al., 2017). We chose to focus on age, in part due to data availability, but care should be taken when interpreting the results to understand that this is only one dimension of vulnerability.

While it is important to consider these limitations, several studies have found good correspondence between areas of high risk, as estimated by the coastal vulnerability model and empirical data on impacts from coastal hazards (Arkema et al., 2013; Cabral et al., 2017). In the United States the states with the most at-risk populations as estimated by the model were also those states with the highest number of fatalities from storms over a 10-year time period (Arkema et al., 2013). In Mozambique, the most exposed districts tended to experience the greatest damages and human fatalities over a 30 year period (UNISDR, 2016; Cabral et al., 2017). And in this study, two of the areas most impacted by Hurricane Matthew – the southern coast of New Providence and the northern tip of Andros – are some of the highest risk locations according to modeled results (**Figure 6**, see Lowe Sound in Discussion, Caribbean Disaster Emergency Management Agency [CDEMA], 2016a,b,c,d; Pacific Disaster Center [PDC], 2016). These comparisons lend weight to our results for The Bahamas and in general indicate that the coastal vulnerability model is a robust, yet methodologically simple approach which can be applied even in data-scarce areas to help decision-makers understand where nature-based solutions may be feasible in their region under different conditions.

CONCLUSION

This study demonstrates an accessible approach and tools to produce a spatially explicit national-scale assessment of coastal hazard risk for The Bahamas, and to identify where coastal and marine habitats provide protection to vulnerable communities, now and under future SLR. We found that coastal and marine ecosystems in The Bahamas play a substantial role in protecting communities from coastal hazards, and will become more important as sea levels rise. The results were used to support on-the-ground efforts to build coastal resilience in The Bahamas and to provide training and build capacity for Bahamians from key sectors. The modeling approach and assessment we report on here also has value for international initiatives, especially as new technologies make coordinated data collection and sharing more feasible across the globe (United Nations Office for Disaster Risk Reduction [UNDRR], 2019). Our methods could be useful to UN member states, which through the Sendai Framework (UNISDR, 2015), aim to reduce disaster risk and losses of lives, livelihoods, and infrastructure. As another example, our results highlight approaches for operationalizing the Sustainable Development Goals through leveraging ecosystems for climate mitigation, adaptation, and coastal resilience. Importantly, our study serves as an encouraging example for other regions and countries seeking to assess and implement nature-based approaches to risk reduction for vulnerable coastal populations.

DATA AVAILABILITY

The datasets generated for this study are available at https://doi.org/10.5281/zenodo.3387713.

AUTHOR CONTRIBUTIONS

KA and MR conceived the research. JS, RG, SS, SHM, SM, KW, GV, and KA prepared the data and conducted the analysis. JS and KA prepared the manuscript with contributions from SM, SHM, SS, AT, BL, ML, MR, KW, GV, and RG. GV and KA developed the hazard index. GV, with guidance from JS, created an online portal to host the results, which was used in training. BL, ML, SHM, and AT provided guidance, vetted the results, and were key partners in the decision-making processes discussed in this manuscript.

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Managing Marine Protected Areas in Remote Areas: The Case of the Subantarctic Heard and McDonald Islands

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Large marine protected areas (MPAs) are increasingly being established to contribute to global conservation targets but present an immense challenge for managers as they seek to govern human interactions with the environment over a vast geographical expanse. These challenges are further compounded by the remote location of some MPAs, which magnify the costs of management activities. However, large size and remoteness alone may be insufficient to achieve conservation outcomes in the absence of critical management functions such as environmental monitoring and enforcement. The Australian subantarctic Heard Island and McDonald Islands (HIMI) Marine Reserve is among the world's most remote MPAs with notoriously harsh oceanographic conditions, and yet the region's rich mammal and fish resources have been exploited intermittently since the mid-1800s. More recently, the development of lucrative international markets for Patagonian toothfish, sold as Chilean seabass, led to the growth in both legal and illegal fishing. In 2002, to conserve the unique ecology and biodiversity in the area, Australia declared a 65,000 km2 MPA around HIMI. Worldwide, government agencies have, however, struggled to develop cost-effective institutional arrangements for conservation. This paper therefore draws upon the social-ecological systems meta-analysis database (SESMAD) to characterize the structure of conservation governance and outcomes in the HIMI Marine Reserve. The Marine Reserve has generally been successful in supporting a sustainable fishery while addressing threats to biodiversity. The remote and isolated nature of the Marine Reserve was critical to its success, but also benefited greatly from collaborations between managers and the fishing industry. Commercial fishers keep watch over the Reserve while fishing, report any observations of illegal fishing (none since 2006/07), and have at times been asked to verify remote observation of potential illegal fishing vessels. The industry also undertakes annual ecological surveys in the MPA, allowing managers to track environmental trends. The fishing industry itself highlights the importance of industry participation in conservation planning, strengthened by secure access to resources via statutory fishing rights, which provide critical incentives to invest in conservation. We therefore reflect on the potential application of this case to other remote large MPAs, highlighting potential directions for future research.

Keywords: conservation, common pool resources, marine protected areas, toothfish, subantarctic, Southern Ocean, collaboration, participation

INTRODUCTION

Large marine protected areas (MPAs) are increasingly being established to contribute to global conservation targets (e.g., Gruby et al., 2016), but present an immense challenge for managers as they seek to govern human interactions with the environment over a vast geographical expanse (Wilhelm et al., 2014). These challenges are further compounded by the remote location of some of these MPAs, which result in rapidly rising costs for a range of governance activities, including environmental monitoring and enforcement (Jones and De Santo, 2016). Nonetheless, environmental monitoring and enforcement are fundamental to sustainable environmental governance (Ostrom, 1990; Cox et al., 2010), even in remote areas (Agnew et al., 2009; Muir, 2010) where advances in technology and lucrative resources compel actors to exploit opportunities at the few remaining frontiers of human society (Watson et al., 2015; Tickler et al., 2018). As a result, there is a growing need to better understand strategies for governing large and remote MPAs to protect their unique ecological features and species of conservation concern.

The Australian-governed subantarctic Heard Island and McDonald Islands (HIMI) Marine Reserve, located more than 4,000 km from major human populations (**Figure 1**), is among the world's most remote MPAs. HIMI are among the least disturbed islands in the world and the least impacted islands in the Southern Ocean (e.g., minimum alien species) (IUCN, 2017; Whinam and Shaw, 2018). Heard Island is also one of the only subantarctic islands with a continuously active volcano. HIMI support large breeding populations of marine birds and mammals, and the surrounding waters are prime foraging areas for a number of marine predators that also rely on the land for part of their life-history, including threatened seals and albatross, an endemic cormorant, and four species of penguins (Green and Woehler, 2006; IUCN, 2017). The marine region supports a range of slow-growing and vulnerable benthic organisms (e.g., cold-water corals and sponges), several endemic fish and benthic species, and nursery areas for a range of fish species, including Patagonian toothfish (Dissostichus eleginoides) (Meyer et al., 2000; Duhamel and Welsford, 2011; Welsford et al., 2019).

Despite its remoteness and notoriously harsh oceanographic conditions, the region's rich mammal and fish resources have attracted harvesters since the mid-1800s (Downes and Downes, 2006). More recently the development of lucrative international markets for Patagonian toothfish, sold as Chilean seabass, led to the growth in both legal and illegal fishing around HIMI (Patterson and Skirtun, 2012). In 1997, it was estimated that approximately 70 illegal fishing vessels were operating in the Southern Ocean, and could earn up to a million dollars on a single trip (Baird, 2004).

In 2002, to conserve the unique ecology and biodiversity in the area, Australia declared a 65,000 km2 no-take marine reserve around HIMI (Welsford et al., 2011) (**Figure 2**). Yet due to the remoteness of this area, Australian Government agencies have faced the difficult task in devising cost-effective institutional arrangements for its conservation and management. While these volcanic islands with their rich populations of birds and mammals have attracted scientists since their discovery, the logistics of operations there have proved difficult (Green and Woehler, 2006). A national scientific base was established on Heard Island in 1947, but was abandoned by 1955 (Munro, 2006). Since then scientific operations have been sporadic, with only two dedicated scientific expeditions to the HIMI Marine Reserve since it was designated (in 2003/04 and 2016; Green and Woehler, 2006; AAD, 2019).

Management of the Reserve, including activities such as enforcement, monitoring and research, are a significant challenge for all stakeholders. Indeed, leading up to the time that the MPA was declared, there were growing concerns about illegal, unreported, and unregulated (IUU) fishing in the area, which precipitated an investment of more than AUD $10 million to enhance patrols in the HIMI waters, and where a single trip could cost upwards of AUD $2 million (Baird, 2004). Given the high costs and intermittent nature of funding for enforcement and research, the fishing industry has played an important role in addressing these gaps and contributing to efforts to reduce or minimize threats to biodiversity.

This paper presents a case study of the HIMI Marine Reserve, and the role that legal toothfish fishers have played in its management from the establishment of the Reserve in 2002 until 2012 (a 10-year snapshot). The HIMI Marine Reserve was expanded on 29 March 2014 (to 71,000 km2; with a new Management Plan), the impacts of which are beyond the scope of the current study. Here we build upon a broader effort to systematically code and analyze the design and performance of large-scale MPAs around the world (Ban et al., 2017; Davies et al., 2018). The remainder of this paper is organized in the following way. First, we briefly describe the methods that were used to code and analyze the HIMI Marine Reserve. We then provide a brief history of HIMI as it transformed from a temporary base for sealers in the 1850s onward to one of the world's largest no-take marine reserves. This is followed by an analysis of the critical role that fishers have played in its development, implementation, and performance. We then conclude with a brief discussion about potential insights for

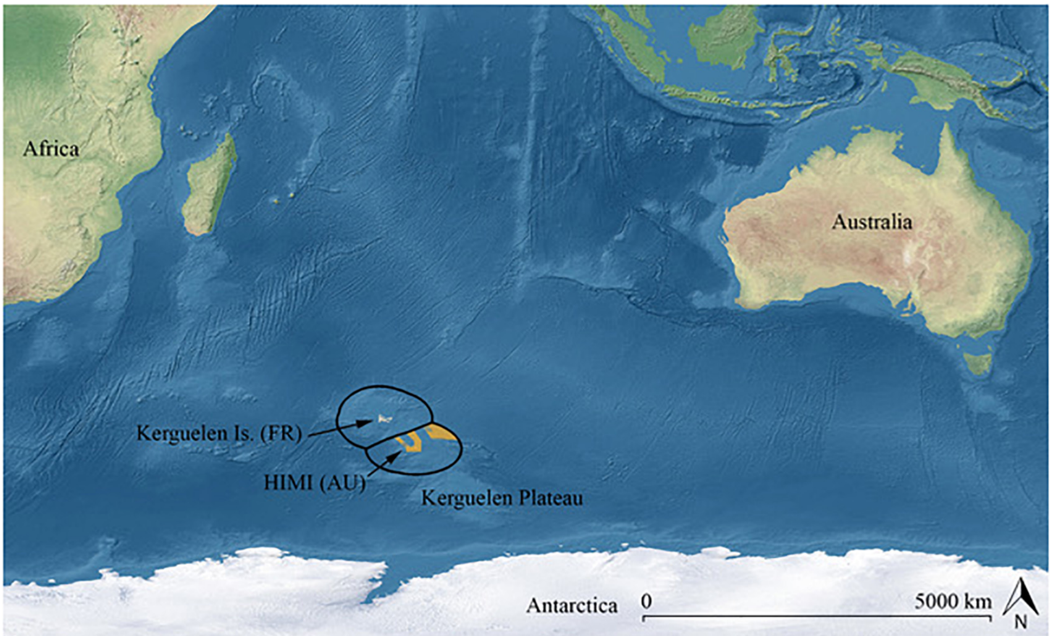

FIGURE 1 | The remote Heard Island and McDonald Islands (HIMI). HIMI are small subantarctic islands on the Kerguelen Plateau located ∼4,000 km southwest of Australia and ∼1,600 km north of Antarctica. The original HIMI Marine Reserve (65,000 km2) is shown in orange (note that the boundaries of the MPA were expanded in 2014). The Australian governed HIMI is adjacent to the French Kerguelen Islands. Australia's Exclusive Economic Zone (EEZ) and the adjacent French EEZ, are illustrated by the black circular lines.

the design and implementation of MPAs in other remote areas around the globe.

MATERIALS AND METHODS

We performed a longitudinal, qualitative, case study (Yin, 2014) of the governance of the HIMI Marine Reserve. A social-ecological systems framework (Ostrom, 2009; Cox, 2014) was used to structure the analysis by identifying key components (resources, actor groups, governance system) of the HIMI Marine Reserve and coding the attributes of those components as part of the collaborative Social-Ecological Systems Meta-Analysis Database (SESMAD) project (Cox, 2014). Through an online platform, SESMAD facilitates systematic collection of information on the social and ecological attributes of large-scale social-ecological systems, the basic unit of analysis, through content analysis of secondary data (e.g., published studies, gray literature) and primary data (e.g., interviews). The SESMAD database provides a simple, and yet powerful approach for systematically coding and analyzing cases through interactions among three core components.

In the HIMI case, we systematically coded (i.e., categorized or indexed) (Saldaña, 2015) variables within the SESMAD database, drawing on extensive peer-reviewed and gray literature to develop an understanding of relevant resources, actors, and the governance systems that influence their interactions with the environment. We analyzed the case between 2002 and 2012, which reflects the establishment of the Reserve in 2002 and our reliance upon secondary data, which often results in a lag between data collection and their broader availability for review. This is consistent with the SESMAD approach and previously published studies using these methods (e.g., Fleischman et al., 2014; Ban et al., 2017). We focused on peer-reviewed studies, and reports and other documentation (policy, legislation, management plans) published by agencies involved in the management of HIMI. We also carried out multiple interviews with three key informants to validate our coding and illuminate important details about governance processes, including the role of different agencies in the management of the HIMI Marine Reserve. We selected participants based on their in-depth experience in research and management of the Marine Reserve. Our study was approved by the University of Victoria's Human Ethics Research Board (ethics protocol number 14-118), and we obtained informed consent from all participants.

We focused on three types of environmental commons in coding the HIMI case study: Patagonian toothfish (Dissostichus eleginoides; the main fishery in the region; environmental commons 1), king penguin (Aptenodytes patagonicus) as an ecosystem indicator (best studied bird in the area, sensitive to climate and environmental changes; environmental commons 2), and light-mantled sooty albatross (Phoebetria palpebrata; long-term presence on HIMI; environmental commons 3) as a migratory species indicator. Two governance systems and three actor groups were also included. The HIMI Marine Reserve Management Plan (governance system 1) governs the land and ocean within the Australian exclusive economic zone (EEZ) around HIMI and is implemented by the Australian

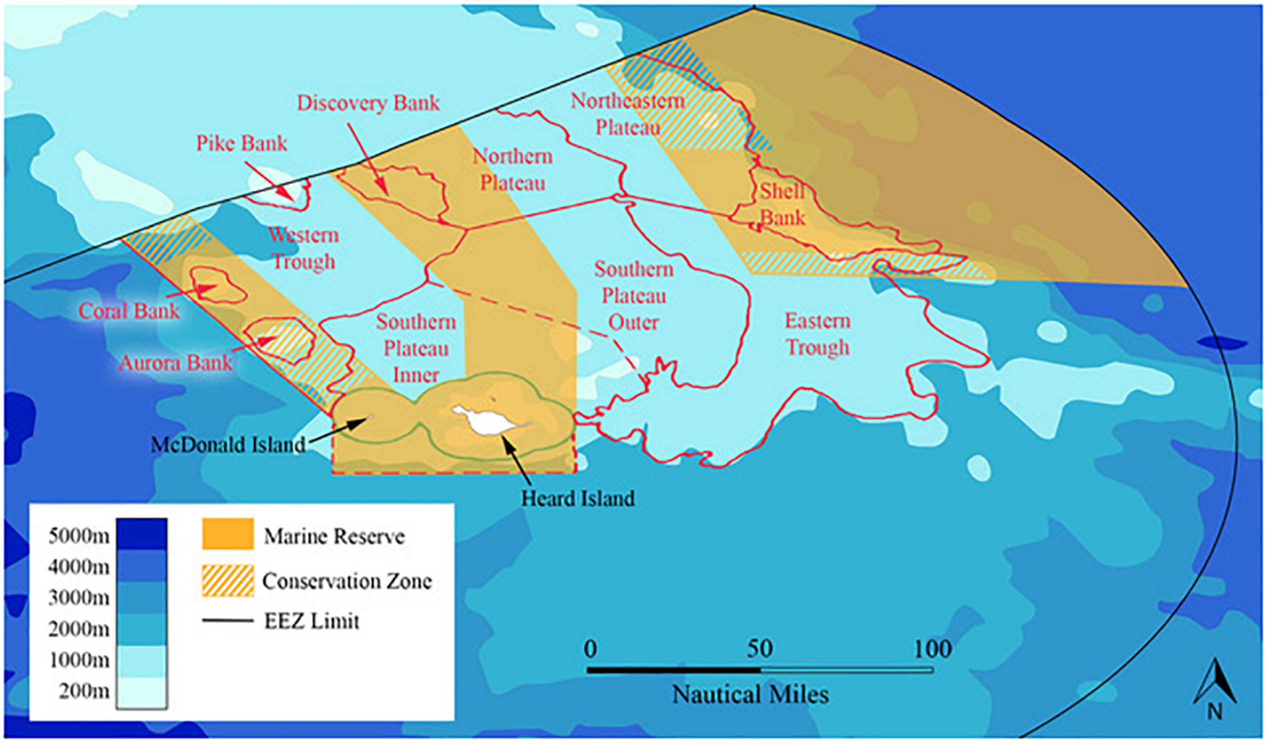

FIGURE 2 | Heard Island and McDonald Islands (HIMI) Marine Reserve and bioregions. Islands represented by white features. Bathymetry (1,000 m increments) is represented by blue coloration (with shallower areas being lighter in color). Red boundaries represent physical units underpinning the marine reserve proposal (based on Meyer et al., 2000). The 2002 HIMI Marine Reserve (65,000 km2) is shown in orange, with no-take areas represented by solid areas and Conservation Zones represented by the hatched areas. Conservation areas underwent further study before some of these regions were incorporated in an expanded HIMI Marine Reserve in 2014 (see text). The HIMI exclusive economic zone boundary is shown by the black line.

Antarctic Division (actor 1). The HIMI Fishery Management Plan (governance system 2), meanwhile, regulates the harvest of toothfish and icefish resources by fishers (actor 2) within the EEZ and is implemented by the Australian Fisheries Management Authority (actor 3). The content of the HIMI MPA case study is publicly available at https://sesmad.dartmouth.edu/ses_cases/18.

RESULTS

Heard Island and McDonald Islands Marine Reserve: Background

Heard Island and McDonald Islands (HIMI) are remote volcanic islands in the South Indian Ocean. Located in one of the most isolated regions of the world, the islands are 1,500 km north of Antarctica and about 4,000 km from Australia, South Africa, and Madagascar (**Figure 1**). The uninhabited islands were discovered in 1853 by American Captain John Heard and were used intermittently as a sealing site between 1856 and the 1880s (Downes and Downes, 2006), with occasional visits by scientific researchers (Green, 2006). By the 1880s, seal populations were decimated, largely ending sealing operations (Downes and Downes, 2006). No nation state claimed HIMI until 1910 when the United Kingdom formally established a claim (Green, 2006). In 1947, with the establishment of an Australian research station on Heard Island, the United Kingdom transferred administration and control of the Islands to the Australian Government (Green, 2006). At that point the islands became governed by Australia as an Australian External Authority through the Heard and McDonald Islands Act of 1953 (Goverment of Australia, 1953).

The Heard Island research station was abandoned in 1955 due to the difficulty and expense of maintenance and operations, and because of the Australian government's priority to support its new Mawson base on the Antarctic continent on the coast of Mac. Robertson Land (Munro, 2006). Since then the islands have been visited only sporadically for research or management (Green and Woehler, 2006; AAD, 2019). Visits by tourists are also only sporadic (see e.g., Heritage Expeditions, 2018). Currently, the most frequent visitor to the area are commercial fishers, which annually target Patagonian toothfish (Dissostichus eleginoides) and mackerel icefish (Champsocephalus gunnari) in the waters around the islands (AFMA, 2018).

In 1979, Australia declared a 200-nautical mile fisheries zone, which in 1994 changed to an official EEZ, abutting France's subantarctic Kerguelen Islands EEZ (Goverment of Australia, 1979, 1994) (**Figure 1**). HIMI also falls within the governance boundaries of the 1980 Convention on the Conservation of Antarctic Marine Living Resources (CCAMLR), of which Australia is a signatory (CCAMLR, 1980). The 1991 Australian

Fisheries Management Act regulates all fishing within the HIMI EEZ (Goverment of Australia, 1991).

Australia's commitment under the Convention on Biological Diversity (CBD, 1992) led Australia to develop a National Strategy for the Conservation of Australia's Biological Diversity, which included an objective to develop a national representative system of marine reserves (Goverment of Australia, 1996). In 1996, HIMI (with a 12 nm buffer portion of the surrounding waters) were declared a Wilderness Reserve by Australia (AAD, 1995). In 1997, the islands were then added to the World Heritage List (UNESCO, 1997). During this time, the land and 12 nm ocean portion of the Wilderness Reserve were also managed as an IUCN Category 1a nature reserve (AAD, 1995). In 1998, Australia released their National Oceans Policy, which identified HIMI as one of the five priority areas for inclusion in a national representative system of MPAs (Goverment of Australia, 1998). In 1999 a Strategic Plan for a National System of MPAs was developed (ANZECC TFMPA, 1999). Simultaneously, the Environmental Protection and Biodiversity Conservation Act 1999 entered into force, providing a legal process for establishing and managing marine reserves (Goverment of Australia, 1999). The Federal Government's Environment Australia commissioned the Australian Antarctic Division to complete a comprehensive compilation and review of the conservation values in the marine environment around HIMI (Meyer et al., 2000).

A comprehensive review of the existing geophysical, oceanographic and biological data of the marine region identified 13 distinct physical units within the HIMI EEZ based on a range of physical variables (e.g., bathymetry, sediment characteristics, water temperature, salinity, currents) (Meyer et al., 2000) (**Figure 2**). The proposed reserve design was generally consistent with conservation design principles of being comprehensive, adequate and representative, including a portion of almost all biophysical units. Efforts were made to include areas used by land-based breeding predators, and to provide some connectivity between areas (e.g., to allow juvenile fish migration from shallow nursery to deeper areas) (Welsford et al., 2011). The reserve was also designed with the explicit intent of providing long-term protection in the event of changes in the distribution of species due to climate change (Welsford et al., 2011).

Based on this proposal, after comprehensive stakeholder consultation (described below) a 65,000 km2 HIMI Marine Reserve and Conservation Zone was subsequently established in 2002 (AAD, 2018) (**Figure 2**). The HIMI Marine Reserve Management Plan was developed and entered into force in 2005, establishing rules and regulations for human activities within the Reserve. The Management Plan is administered by the Australian Antarctic Division, but the Division works in collaboration with multiple agencies and other stakeholders – especially the fishing industry – in undertaking research, monitoring, and enforcement (Goverment of Australia, 2005) (**Tables 1–3**).

The main purpose of the MPA is to protect: the conservation values of HIMI, including the World Heritage and cultural values; biodiversity; the unique features of the benthic and pelagic environments; representative portions of the different marine habitat types; and marine areas used by land-based marine predators for foraging activities (Goverment of Australia, 2005) (**Table 2**). The MPA is managed as fully no-take. However, fishing, which has occurred since 1997 for toothfish and icefish is allowed in the waters adjacent to the Reserve (Goverment of Australia, 2005).

TABLE 1 | Collaborative management of HIMI Marine Reserve.

Stakeholder/Agency	Role
Australian Antarctic Division	Main management agency
Australian Fisheries Management Authority (in collaboration with others, including the Subantarctic Management Advisory Committee, the Subantarctic Resource Assessment Group, and the Commission for the Conservation of Antarctic Marine Living Resources).	Involved in fisheries and ecosystem management, research and monitoring
Commercial fishers (Austral Fisheries Pty Ltd and Australian Longline Pty Ltd)	Integral to fisheries and ecosystem research and monitoring, IUU deterrent and monitoring
Australian Maritime Safety Authority	Liaise on safety issues
Australian Border Force	Patrolling for IUU fishing, monitoring and enforcement, invasive species issues
Tourist and Recreational Visitors	Opportunistic research and monitoring
French National Authorities	IUU monitoring

Stakeholders and agencies involved in managing the HIMI Marine Reserve and their role in management. IUU refers to illegal, unregulated and unreported fishing.

MPA Performance

Fisheries Outcomes

The HIMI Marine Reserve contributes to the sustainability of the toothfish fishery by protecting aspects of toothfish life history, connectivity and providing opportunities for regular research and monitoring (Meyer et al., 2000; Goverment of Australia, 2002). While currently both icefish and toothfish are harvested in the waters around HIMI, we focus on toothfish since they sustain the largest fishery (Patterson and Skirtun, 2012). While toothfish populations have decreased from about 82% of unfished levels in 2002 to 62% in 2012 (CCAMLR, 2013), this is consistent with the goals of the fishery management plan (AFMA, 2002) and Southern Ocean management thresholds adopted by CCAMLR (Constable et al., 2000). Both fisheries have been certified as sustainable by the Marine Stewardship Council (icefish since 2006; toothfish since 2012) (MSC, 2018) and are considered precautionary and sustainable by Australian Government agencies (Constable and Welsford, 2011; Patterson and Skirtun, 2012; AFMA, 2014).

Fishing regulations are strictly enforced through several monitoring and reporting mechanisms. These include two independent onboard observers, vessel and port monitoring systems, the Australian Fisheries Management Authority or Australian Defense Force patrols, and CCAMLR reporting (AFMA, 2002). Fishers face significant government sanctions for violating rules (including fishing within the MPA) and risk losing their highly coveted Marine Stewardship Council certification, resulting in high levels of compliance (see e.g., AFMA, 2014;

TABLE 2 | Collaboration toward meeting primary management goals.

Management Goal	AAD	AFMA or Fishers	Others
Zoning and IUCN Category (land)	**Active:** via zoning		
Environmental Assessment and Approval (for HIMI visitors/activities)	**Active:** environmental impact assessments required (e.g., land visitors); applications to enter Marine Reserve (e.g., research vessels)		
Visitor Management and Reserve Use	**Passive:** no visitors during snapshot		
– *Access and Transport*			
– *Management of Facilities (land)*	**Passive:** no visitors during snapshot		
– *Visitor Management and Commercial Activities*	**Passive:** no visitors during snapshot		
– *Communicating Reserve Values*	**Active:** via websites		
Natural Heritage Management – *Flora and Fauna*	**Mostly Passive:** little to no data on conservation status for many target fauna (e.g., seabirds, mammals); Some assistance from fishery	Assistance with some wildlife conservation issues (e.g., mitigating seabird bycatch)	
– *Natural Asset Use*	**Active:** in partnership	Assistance with ensuring no fishing in Reserve	Border Force and French authorities assist with monitoring for fishing activities
– *Waste Management*	**Passive:** no visitors during snapshot		
– *Prevention and Management of Alien Species and Disease*	**Passive:** no visitors during snapshot		
– *Research and Monitoring*	**Active:** largely in partnership with the fishery	Commercial fishers highly involved with research and monitoring (**Table 3**)	
Cultural Heritage Management	**Active:** Communication goals/prescriptions		
Stakeholders and Partnerships	**Active:** in partnership		
Business Management – *Operational Management*	**Active:** in partnership		
– *Compliance and Enforcement*	**Active:** in partnership	Assistance from fishery	Assistance from Border Force and French authorities
– *Financial Management*	**Active:** administrative		
– *Emergency Management*	**Passive:** no visitors during snapshot, but plans in place		
Performance Assessment	**Active:** research that led to conservation zone inclusion (2014 addition)		

Main management goals of the 2005 Heard Island and McDonald Islands (HIMI) Marine Reserve Management Plan (left column) indicating the responsible agency and mechanisms for achieving each goal (center column and right columns). Passive management indicates that the management goal is likely being met, but not by active management by agencies (de facto by no activity or already existing activities). AAD refers to the Australian Antarctic Division. AFMA refers to the Australian Fisheries Management Authority. Note that the HIMI management plan governs the islands and the surrounding Marine Reserve. Empty cells indicate no involvement.

MSC, 2018). With secure access rights, it is also in the fishers' long-term interest to ensure a sustainable and well managed fishery.

While the toothfish populations currently appear sustainable, the fishery operates in a context of significant uncertainty. For instance, there is growing evidence from genetic studies (Appleyard et al., 2002, 2004), parasite faunal analysis (Brickle et al., 2005) and tag recapture studies (Williams et al., 2002; Duhamel and Welsford, 2011) that suggests that HIMI toothfish are part of a larger Kerguelen Plateau/South Indian Ocean population. Recent stock assessments are beginning to incorporate movement between the HIMI and Kerguelen Island regions (WG-FSA, 2017; Ziegler and Welsford, 2019). Further questions, meanwhile, relate to the habitats and locations used for spawning and larval stages, the exact timing of spawning, the proportion of the population that spawns (i.e., evidence of skip-spawning) (Welsford et al., 2012; Péron et al., 2016). From what is known, toothfish are capable of supporting small-scale fisheries, but due to their life history (slow growth, later age at maturity, long-lived) and relatively small populations (Collins et al., 2010), they are vulnerable to overexploitation. For instance, several populations in the region were heavily overexploited by IUU fishers in the 1990s and early 2000s and have yet to recover (McKinlay et al., 2008; Collins et al., 2010; Welsford, 2011).

Toothfish populations in the circumpolar subantarctic region, including HIMI, were subject to extensive IUU fishing from the mid-1990s to early 2000s (Österblom and Sumaila, 2011). However, as a result of efforts by a variety of stakeholders, including the Australian government and the fishing industry (Österblom and Sumaila, 2011; Österblom and Bodin, 2012), there have been no observations of IUU fishing around HIMI since 2005 (AFMA, 2014), and no sightings of IUU vessels in the CCAMLR Area since 2015/16 (CCAMLR, 2017b). Austral Fisheries, an Australian commercial fishing company, were particularly instrumental in recognizing the environmental and economic threats posed by IUU fishing and spent more than $2 million USD in 2002–2003 on lobbying, surveillance, and hiring

TABLE 3 | Participation of commercial fishery in marine reserve research and monitoring.

Research and Monitoring Priorities	Commercial Fishery Participation
Continuing population counts and monitoring of threatened species to assist in the implementation of the subantarctic Fur Seal and Southern Elephant Seal Recovery Plan, Recovery Plan for Albatrosses and Giant Petrels and Draft Recovery Plan for 10 species of seabirds	Observer counts and species identification of seabirds; reporting requirements on any death, injury, or interaction with vessel or gear
Research and Monitoring toward other recovery plans, action plans, and threat abatement plans	Input at Resource Assessment Group and other advisory committee levels; Assistance in preparing potentially successful approaches
Comprehensive surveys of indigenous species to provide baseline information against which to compare human-introduced or otherwise newly colonized terrestrial, freshwater and marine species	Marine species from random stratified trawl survey; also data collection from two fisheries observers; project based research programs
Long-term whole of reserve and colony specific monitoring to provide fundamental data on the distribution, abundance and population trends of seal and seabird species, with particular emphasis on listed threatened species	Fisheries observers conduct counts from vessel while fishing
Surveys to increase knowledge of the biodiversity of the reserve, and its response to current conditions and climate change.	Annual random stratified trawl survey; benthic assemblages sled project, benthic assessment camera work
Hydrographic surveys for producing and updating of marine charts.	Bathymetric data from fishing operations granted upon request (with confidentiality clauses in place); in collaboration with AAD, Universities and Geosciences Australia
Opportunistic monitoring of the distribution of cetaceans during AAD expeditions, by fishing vessels, yachts, tourist vessels, merchant vessels, spotter aircraft	Active monitoring in collaboration with AAD, AFMA observers, Australian and French patrols, scientists, and (occasional) tourist vessels.
Acoustic mapping of the substratum	Active mapping in collaboration with AAD and Universities
Stratified random sampling of the benthos, particularly habitat-forming benthos such as sponges and corals, to determine the extent of differences in the assemblages and habitats between the biophysical units used to develop the reserve	Active sampling in collaboration with AAD and AFMA observers
Stratified random sampling of benthos within and outside the reserve, to determine how well the reserve configuration protects the features it was designed to protect	Active sampling in collaboration with AAD and AFMA observers
Stratified random sampling within and outside the reserve of target species in the HIMI fishery	Active sampling in collaboration with AAD and AFMA observers
Research into the impacts of commercial fishing in adjacent waters on the reserve and/or its key components (e.g., protected species)	Active research via trawl survey, AFMA observers, data collection from vessels, reporting requirements, advisory committees
Monitoring changes in the degree to which anthropogenic threats affect threatened animal species	Some research on environmental variability and some research and management to ensure minimal anthropogenic threats of fishing on seabirds, fish species, ecologically related species
Investigating the cumulative impacts of research programs and other activities on threatened species or species and their habitats that are vulnerable to human disturbance	Ongoing as research programs are undertaken
Fish stock assessments	Substantive involvement with data collection (e.g., from AFMA fisheries observers), participation in advisory committees, and involvement in CCAMLR

HIMI Marine Reserve research and monitoring priorities (Goverment of Australia, 2005) which the commercial fishery participates in (R. Arangio, Austral Fisheries, 29 June 2016; D. Welsford, AAD, 21 October 2016). AAD refers to the Australian Antarctic Division; AFMA refers to the Australian Fisheries Management Authority.

private investigators to identify IUU operators (Österblom and Sumaila, 2011). Austral Fisheries continues to provide support in the form of surveillance, along with the French and Australian governments (R. Arangio, Austral Fisheries, 29 June 2016). They are also an active member of the Coalition of Legal Toothfish Operators (COLTO), a group of 50 toothfish fishing companies and support industry companies from a dozen nations that advocates for legal and environmentally sustainable toothfish fishing operations (COLTO, 2018).

Ecological Outcomes

Habitat assessments have shown that a significant majority of vulnerable organisms occupy the HIMI seafloor at depths of less than 1,200 m, a range that overlaps with the trawl and longline fisheries (Welsford et al., 2014). However, most of the trawling occurs in a relatively small area, which has limited habitat impacts to less than 1.5% of biomass in waters less than 1,200 m (Welsford et al., 2014). Furthermore, the HIMI Marine Reserve contains areas in which 40% or more of the benthic biomass is considered most vulnerable to bottom fishing. However, it has been estimated that only about 0.7% of the seafloor area within the HIMI EEZ has experienced interactions with bottom fishing gear between 1997 and 2013 (Welsford et al., 2014).

Relatively little is known about conservation outcomes for species that rely on the HIMI Marine Reserve. Our analysis focused on two species for which at least some data on their life history and status is available, and which may provide an indicator of ecosystem conditions and role in the life histories of migratory species (Parsons et al., 2008; Einoder, 2009), respectively: king penguin (Aptenodytes patagonicus;

Bost et al., 2013; Cristofari et al., 2018) and light-mantled sooty albatross (Phoebetria palpebrata; Phillips et al., 2016).

King penguins have largely recovered from historical over-exploitations throughout the subantarctic (as an oil source) throughout the region in the late 19th and early 20th centuries (Bost et al., 2013). Populations at Heard Island (as well as Kerguelen) have experienced slower rates of recovery compared to other subantarctic populations, and still appear to be increasing (Woehler, 2006; Bost et al., 2013 and references therein). A 1947 visit to Heard Island, for instance, found only three king penguins, compared to the approximately 80,000 pairs found in 2003/4 (Woehler, 2006). Since then, the available data suggest that the population continues to increase (Heritage Expeditions, 2012; Bost et al., 2013; E. Woehler, BirdLife Tasmania, 28 August 2015); although the lack of a population survey or regular observations since 2003/04 contributes to significant uncertainty about the contemporary population status of king penguins and health of the broader marine ecosystem.

The HIMI Marine Reserve Management Plan addresses a number of threats to king penguins and the marine ecosystem. On land, management zones are used to protect breeding areas, tourists are prohibited from closely approaching and harassing penguins; and scientists require permits to study them (Goverment of Australia, 2005). At sea, meanwhile, some foraging areas fall within the boundaries of the Marine Reserve, but also extend into the French EEZ (around Kerguelen) and into the high seas (see e.g., Meyer et al., 2000). King penguins forage at great depths (reaching 440 m) and feed on pelagic fish, especially myctophids (Moore et al., 1999; Bost et al., 2013). If myctophids are not readily available, king penguins may also feed on mackerel icefish – a species which is also commercially harvested outside the boundaries of the Marine Reserve, thus potentially putting penguins in competition with commercial fishers (Bost et al., 2013). King penguins travel far, especially in the winter (up to 1,800 km from their colony, 5,000 km round trip) (Putz et al., 1999). However, during the breeding season, they typically stay within 500 km of their colonies (Putz et al., 1999). Their foraging ecology has been extensively studied and is strongly dependent on the Antarctic frontal zone features, especially the Antarctic Polar Front (Bost et al., 2015; Cristofari et al., 2018). This makes them highly vulnerable to climate change (Peron et al., 2012; Bost et al., 2013, 2015; Cristofari et al., 2018). Shifts in their main prey, myctophids, are predicted under future climate change scenarios, with unknown consequences for king penguins (Freer et al., 2019).

The MPA was explicitly designed with the intent of protecting breeding sites and foraging grounds for migratory seabirds, including light-mantled sooty albatross (Meyer et al., 2000; Goverment of Australia, 2005). These circumpolar birds can travel more than 6,000 km from breeding sites (including sites on Heard Island) to their foraging grounds (Weimerskirch and Robertson, 1994). Light-mantled sooty albatrosses demonstrate high breeding site fidelity but because they are biennial breeders, they do not return each year (Bonnevie et al., 2012). The population at Heard Island has been estimated at somewhere between 200 and 500 nesting pairs based upon 2000/01 and 2003/4 surveys (Green and Woehler, 2006; Woehler, 2006). This population is relatively stable based on comparisons with early counts from the 1950s which also estimated between 200–500 pairs (Downes et al., 1959). Historical trends and expert interview (E. Woehler, BirdLife Tasmania, 28 August 2015) suggest the population is stable or increasing, the latter being due to the novel nesting sites found since the 1950s (Woehler, 2006). Counts by tourists in 2012 also support estimates of a persistent population (Heritage Expeditions, 2018). However, accessibility and changes in nesting locations pose significant challenges for obtaining a reliable estimate of the breeding population (Woehler, 2006).

The HIMI Marine Reserve Management Plan addresses a number of potential threats to light-mantled sooty albatross on land and sea portions of the Reserve. This includes requirements for visitor permits, restrictive zoning of land areas to concentrate impacts and avoid nesting areas; and prohibitions against fishing in the Reserve (Goverment of Australia, 2005). Protection of land areas has been greatly facilitated by isolation. However light-mantled sooty albatross breeding at HIMI continue to face significant threats emerging from beyond the boundaries of the Reserve. These include climate change and incidental mortality in legal and IUU fisheries for tuna and toothfish (ACAP, 2012; Phillips et al., 2016; BirdLife International, 2018). However, the toothfish fishery at HIMI has proved remarkably successful in avoiding such impacts through the adoption of innovative technologies and mitigation measures (AFMA, 2014). Since 2006, very few birds (1–7 per year) are taken in the toothfish fishery at HIMI, none of which were light-mantled sooty albatross (CCAMLR, 2017a).

HIMI: Factors Contributing to Conservation Success

HIMI has been offered as an example of successful marine conservation in a remote and challenging environment (Constable and Welsford, 2011; Goldsworthy et al., 2016; MSC, 2018). Our case study indicates that the success of the HIMI Marine Reserve stems from two critical factors: (1) remoteness and isolation which reduce human threats and impacts, and (2) collaboration with the fishing industry, which has allowed stakeholders to manage threats posed by the fishing industry and provide an efficient approach for addressing management gaps.

Remoteness

The remoteness of HIMI and the harsh climate it experiences have made significant contributions to the protection of biodiversity on land and marine areas, by limiting direct human interactions with the environment since sealing and whaling activities ceased in the early 20th century (Green and Woehler, 2006; IUCN, 2017; Whinam and Shaw, 2018). Since the 1960s, Heard Island has experienced mostly sporadic visits from scientists and tourists, while McDonald Island has only been visited on two occasions (AAD, 2018). While isolation offers significant protection from a number of threats, it also poses significant challenges for managing the Reserve and responding to emerging threats (Whinam and Shaw, 2018). A lack of funding and logistical support by the Australian Antarctic Division and the high costs of traveling to HIMI have prevented managers from undertaking activities specified in management plans, such as ecological monitoring which could provide important details about the status and trends for species of conservation concern (see **Tables 2**, **3**).

The remoteness and difficulty of access also means that managers know very little about the status of marine life, with the exception of targeted commercial fish species, around HIMI (IUCN, 2017; **Tables 2**, **3**). Similarly, satellite imaging of Heard Island has revealed significant glacial retreat (see e.g., Mitchell and Schmeider, 2017; AAD, 2018), but scientists and managers currently lack an understanding of the potential impacts of these changes (and other climate change impacts) on birds and mammals on the island (Chambers et al., 2013, 2014). Climate change has caused phenological changes in many other Southern Ocean seabirds, especially penguins and some albatrosses, including species that live on HIMI (Chambers et al., 2013, 2014). Finally, although there is no indication that climate change has adversely affected HIMI toothfish populations as of yet, toothfish recruitment may be sensitive to changes in sea surface temperature and could be affected by predicted future changes (Trathan and Agnew, 2010; Constable et al., 2014).

Collaboration With the Fishing Industry

Australia adopted a highly transparent and collaborative process for developing the HIMI MPA, including opportunities for significant participation by the fishing industry. After reviewing ecological values in the area and proposing an MPA design that followed best practices in conservation (Meyer et al., 2000), the Australian Antarctic Division released the proposal in early 2001 and began an extensive (18-month) consultation process which included the formation of the HIMI stakeholder group (Welsford et al., 2011; Goldsworthy et al., 2016). This group included members from the policy and research branches of the Australian Antarctic Division, the fishing industry and a variety of non-governmental organizations. After consultation, the stakeholder group largely supported the design and rationale for the MPA proposal and they supported inclusion of approximately 85% of the original proposal (Welsford et al., 2011). The HIMI stakeholder group, chose to temporarily set some of the proposed areas as "Conservation Zones" which allowed for further research on the conservation values of these areas against the representativeness of other areas in the MPA as well as for examining the threat of fishing to the conservation values in this area against the economic importance of the fishery (Welsford et al., 2011). The HIMI Marine Reserve was subsequently established in 2002 as a 65,000 km2 no-take (IUCN category Ia) MPA (**Figure 2**), and parts of these conservation zones were incorporated into the expanded Marine Reserve in 2014.

This transparent process resulted in strong support by the fishing industry, which consists of only two companies: Austral Fisheries and Australian Longline. In 2003, Austral Fisheries received an award from the World Wildlife Fund for their involvement in the HIMI and the Macquarie Island Marine Reserves (Austral Fisheries, 2018). The fishing companies have strongly supported the HIMI Marine Reserve and believe it contributes to a stronger and more sustainable fisheries management system.

> *"We have a strong belief in the science that underpins the fishery and we know what can happen if it's not managed properly. The end game is a balance between protection and rational use and we supported the MPA because we knew it would protect benthic assemblages, juvenile fish stocks and create broader ecosystem balance"*
>
> *(R. Arangio, Austral Fisheries, 29 June 2016).*

The two toothfish fishing companies hold individual transferable quotas that provide a secure and long-term right to harvest toothfish resources at HIMI. Although there are a number of important exceptions (see e.g., Ban et al., 2009), individual transferable quotas can provide critical incentives to support the long-term sustainability in fisheries (Grafton et al., 2006; Costello et al., 2008, 2010).

The collaboration between the Australian Antarctic Division and the fishing industry early on lent itself to collaborative management. Moreover, the Australian Fisheries Management Authority employs a 'partnership approach' in their fisheries management (Smith et al., 1999). As was exemplified in the HIMI Marine Reserve process, fisheries management in Australia emphasizes stakeholder involvement in all key area of fisheries management, including stock assessment, research priorities, enforcement and decision-making (Smith et al., 1999). In the case of HIMI, while the Australian Antarctic Division and the Australian Fisheries Management Authority (the government agency which oversees fisheries) are separate bodies with separate mandates and management plans, they work very closely in the management of the HIMI Marine Reserve (see e.g., AFMA, 2002; Goverment of Australia, 2005) (**Tables 2**, **3**). The fishing industry also has an agreement to monitor the MPA, which is complemented by a vessel monitoring system and remote surveillance by the governments of Australia and France via satellites. Ultimately, activities occurring within and adjacent to the MPA are actively monitored, and there are no indications of IUU fishing or other prohibited activities occurring within the HIMI EEZ since 2005 (AFMA, 2014).

HIMI Marine Reserve Management

Australia's Antarctic Territories, including HIMI, are managed by the Australian Antarctic Division, which often struggles with limited resources and fiscal constraints that create challenges for research and monitoring in the HIMI Marine Reserve. As a result, the Division has relied heavily on partners, including the fishing industry, to assist in research and monitoring (**Tables 1–3**). Minor assistance is also provided by the Australian Department of Defense, tourists, and French national authorities who actively undertake research and patrols in the Kerguelen and Crozet EEZ (**Table 1**). The Australian Antarctic Division issues permits for the rare visitors, manages flora and fauna, and monitors compliance with fishing regulations. Management of the Reserve is largely passive in the sense that there is a limited human presence beyond fishing (**Table 2**). In the time since the Reserve was established in 2002, there has only been two dedicated science expeditions to the HIMI Marine Reserve – one in 2003/04 and one in 2016. Two private tourism expeditions have visited the Reserve (in 2012 and 2016) and the Australian Antarctic Division has had one management visit (in 2008) (AAD, 2019). Some research and monitoring is done remotely (e.g., via satellites), while the majority is undertaken in collaboration with the fishing industry (**Tables 2**, **3**). Monitoring for fishing activity is undertaken via satellites, through government vessel patrols (in collaboration with the French Government) and in collaboration with the fishing industry. Other organizations provide support in the form of information (e.g., CCAMLR, COLTO), monitoring, and enforcement (e.g., surveillance carried out by the Australian Border Force) (**Tables 1**–**3**).

Institutional Arrangements With the Fishing Industry

Environmental monitoring for the HIMI Marine Reserve and the broader HIMI EEZ takes place in the context of the "fishery assessment plan," a formal agreement between the Australian Antarctic Division with the Australian Fisheries Management Authority that specifies research activities and responsibilities on an annual basis (D. Welsford, AAD, 21 October 2016; R. Arangio, Austral Fisheries, 29 June 2016). Permits for research activities in the HIMI Marine Reserve (including fish surveys) are issued by the Australian Fisheries Management Authority in consultation with the Australian Antarctic Division (Welsford et al., 2011). The fishing industry is primarily responsible for *ad hoc* monitoring via fisheries observers on vessels and for undertaking the annual random stratified trawl survey (see below; D. Welsford, AAD, 21 October 2016). However, apart from research activities and transit, the fishing industry is strictly prohibited from entering the Marine Reserve (Goverment of Australia, 2005; R. Arangio, Austral Fisheries, 29 June 2016).

While the Australian Antarctic Division leads stock assessment work, the fishing industry carries out supportive research and monitoring on an annual basis, the costs of which it is not compensated for. The Australian Fisheries Management Authority policy is that the industry provides in-kind support (equivalent to about $600,000 AUD) for the stratified survey alone (D. Welsford, AAD, 21 October 2016). These are the conditions agreed to for entry into the fishery. Industry also pay for fish tagging (D. Welsford, AAD, 21 October 2016), which includes both the cost of the tag, but also the opportunity cost of the released fish. Two fisheries observers, which are required to be on board at all times, are also funded by industry. Industry may also take a third observer to assist with completing surveys or required research from time to time. Industry costs are generally shared between the two fishing companies as a proportion of the fishing quota holdings (R. Arangio, Austral Fisheries, 29 June 2016).

Fishing Role and Activities

Random stratified trawl survey

Since 1997, commencing with the start of the commercial fishery for toothfish and icefish, the fishing industry has undertaken an annual Random Stratified Trawl Survey, typically occurring in April-May (AFMA, 2014). The survey covers 10 regions (strata) of the Heard Island Plateau that define areas of similar depth and/or fish abundance. The annual surveys have continued since the establishment of the MPA and routinely incorporate stations inside and outside the boundaries of the MPA (Welsford et al., 2011). Approximately 20 days of the industry fishing time is provided to complete the survey (R. Arangio, Austral Fisheries, 29 June 2016). The Australian Antarctic Division provides a specific set of instruments for the survey, in addition to tow times, tow directions, and a list of stations randomly dotted across the plateau. Approximately 15–20% of the 160 stations are found in the MPA (R. Arangio, Austral Fisheries, 29 June 2016). The survey is conducted by Austral Fisheries, on behalf of the two fishing companies that own quota in the HIMI toothfish fishery (Austral Fisheries and Australian Longline) (R. Arangio, Austral Fisheries, 29 June 2016).

Benthic survey

The fishing industry (Austral Fisheries) has also undertaken specific monitoring and survey work to assist the Australian Antarctic Division in past years, including benthic sampling with towed sleds (R. Arangio, Austral Fisheries, 29 June 2016). In 2003 a benthic beam trawl and sled sampling occurred as part of an Australian Antarctic Division and fishing industry funded research project to evaluate the biodiversity inside and outside the Reserve and Conservation Zone. Further work was undertaken in 2007/8 as part of a large collaborative project involving the Division, the fishing industry, the Australian Fisheries Management Authority and the Fisheries Research and Development Corporation, and continued until 2013. The research project involved video habitat monitoring to identify and evaluate benthic assemblages in the HIMI area. Cameras were mounted on trawl gear, longlines, and pots. The video information was combined with habitat mapping and analyses of regional community structures (R. Arangio, Austral Fisheries, 29 June 2016). The resulting study found that more than 98% of habitat was unaffected by fishing and offered further knowledge of the region, including areas within the MPA (Welsford et al., 2014).

Seabird monitoring and technical innovation

While the MPA explicitly includes provisions for migratory species, including foraging areas for albatross (Goverment of Australia, 2005), the primary threat to these birds is incidental bycatch by commercial fishing vessels. In accordance with the HIMI Fishery Management Plan (2002), the fishing industry must implement several seabird bycatch mitigation measures. Internally weighted lines, which are now a global standard for automatic longline fishing vessels allow hooks to sink rapidly out of reach from seabirds (Wiedenfeld, 2016). HIMI fishers use these weighted lines combined with tori lines and brickle curtains on every haul and this combination has minimized seabird interactions (AFMA, 2002, 2014). The fishers also follow restrictions on time of day for setting gear to avoid seabird interactions as well as seasonal closures. The release of offal is prohibited to avoid attracting seabirds to fishing vessels (AFMA, 2002). Each vessel must also have two full time observers (AFMA, 2002). These observers maintain daily records that outline the number and types of seabirds observed while fishing. Further, they are required to report any physical interactions between fishing activities and seabirds (AFMA, 2002, 2014). Australian and New Zealand toothfish fishers have also contributed to the development of innovative technologies designed to reduce threats to seabirds (R. Arangio, Austral Fisheries, 29 June 2016). These seabird bycatch mitigation measures are in accordance with the current scientific consensus and are considered perhaps the best example of seabird bycatch mitigation techniques (see e.g., Croxall, 2008; Wiedenfeld, 2016).

Social monitoring (IUU)

The rapid growth of IUU fishing for toothfish in the late 1990s and early 2000s contributed to the establishment of COLTO (Österblom and Sumaila, 2011). COLTO, along with dozens of other governmental and non-governmental organizations, have worked to dramatically reduce IUU fishing throughout the Southern Ocean, including around the Heard and McDonald Islands (Österblom and Sumaila, 2011). Crew and fisheries observers on commercial fishing vessels have played an important role in these efforts by monitoring for and reporting observations of potential IUU vessels. Crew members report observations directly back to the fishing company (e.g., Austral Fisheries) while the fisheries observer records any vessel sightings and provides this information to the Australian Fisheries Management Authority, the Australian Antarctic Division and CCAMLR (R. Arangio, Austral Fisheries, 29 June 2016). Further, Australia has signed a memorandum of understanding with the French Government for joint patrols and surveillance over the Kerguelen Plateau, which can be undertaken from French or Australian patrol vessels (R. Arangio, Austral fisheries, 29 June 2016). Since 2005, through the joint efforts of the fishing industry, French and Australian Governments, there have been no reports of IUU fishing within the HIMI EEZ (AFMA, 2014).

Fisheries and Fisheries Management Near HIMI

Fisheries in the HIMI EEZ are managed by the Australian Fisheries Management Authority, under the Fisheries Management Act 1991 (Goverment of Australia, 1991) in close cooperation with the Australian Antarctic Division and in accordance with Conservation Measures set by CCAMLR (AFMA, 2014). The HIMI Fishery Management Plan includes the trawl fishery for mackerel icefish and the trawl, longline and pot fisheries for Patagonian toothfish (AFMA, 2002). Longlines were introduced in 2003 and pots were introduced in 2009, though fishing via pots remains at a very low level. The total allowable catch for toothfish between 2002 and 2012 has ranged between 2400–2800 tones (with pots comprising only 30–68 tons) (CCAMLR, 2017a). The toothfish fishery has gradually shifted from trawls to longline (e.g., in 2012 about half the total allowable catch was caught via trawl, but by 2017 it was only 24 tons) (AFMA, 2014) as innovations in longline technology have reduced threats to seabirds (AFMA, 2014; CCAMLR, 2017a). In addition, the Antarctic Marine Living Resources Conservation Act 1981 (Goverment of Australia, 1981), administered by the Australian Antarctic Division, implements Australia's international obligations under the Commission for the Conservation of Antarctic Marine Living Resources (AFMA, 2014).

With regards to Patagonian toothfish, the HIMI Fishery Management Plan establishes rules for setting catch limits, granting fishery quotas, and implementing other fisheries and environmental measures (e.g., gear restrictions, bycatch rules) (AFMA, 2002). The plan is implemented primarily by the Australian Fisheries Management Authority who cooperates with the Australian Antarctic Division to avoid potential impacts on the MPA and ensures consistency with CCAMLR Conservation Measures. The Australian Fisheries Management Authority aims to maintain toothfish populations at sustainable levels, while also attempting to avoid impacts on the broader ecosystem through limits on bycatch and mitigation measures to avoid interactions with seabirds (AFMA, 2002, 2014).

The HIMI fishery is managed using transferable quotas, which are currently held by two Australian fishing companies: Austral Fisheries (71% of fishing rights) and Australian Longline (29% of fishing rights) (R. Arangio, Austral Fisheries, 29 June 2016). Until the 2011/12 season, three or four vessels were in operation per season at HIMI (CCAMLR, 2018). Through consultative fora, toothfish fishers play an active role in the governance and management of toothfish (e.g., through industry representatives at SouthMAC – the Subantarctic Fisheries Management Advisory Committee and SARAG – the Subantarctic Resource Assessment Group – which includes representatives from the fishing industry, conservation groups, scientists and other relevant experts) (AFMA, 2014). Based on advice from SARAG, SouthMAC recommends catch rules to AFMA. Mechanisms are also in place to manage perceived or actual conflicts of interest by members of these groups when they are developing their advice. These fora, along with engagement via COLTO, provide mechanisms for conflict resolution and building trust through repeated face to face interactions (Ostrom et al., 1994).

DISCUSSION

The HIMI Marine Reserve and the role that toothfish fishers have played in its establishment, implementation and management, and its success in managing threats and supporting conservation efforts are a remarkable example of the benefits of participatory conservation planning. The toothfish fishers have made significant efforts to develop technologies and adjust operations to reduce seabird bycatch, and have made a number of critical contributions to the governance of the Reserve, including surveillance and environmental monitoring. This case clearly demonstrates the potential value of adopting participatory conservation models that view resource users not only as a potential threat to the environment, but also as a critical partner for achieving conservation goals (Stoll-Kleemann and O'Riordan, 2011; Andrade and Rhodes, 2012). Toothfish fishers were engaged in the early stages of conservation planning, their input was respected and incorporated in the form of temporary conservation zones, and as a result the fishers have continued to support the Reserve through a range of activities and actions. The HIMI Marine Reserve presents a potentially valuable model that can inform conservation planning, although important questions remain concerning the contexts in which similar approaches are more (or less) likely to prove effective. In particular, the success of participatory conservation planning at HIMI may have been facilitated by a number of critically important enabling conditions that contributed to its success.

First and perhaps foremost, the HIMI Marine Reserve was established in the context of political debates surrounding IUU fishing of toothfish and the potential impacts of Antarctic fisheries in general on what is seen by many as a 'pristine' environment (Potts and Haward, 2006; Stokstad, 2010; Cavanagh et al., 2016). These debates and the potential threats they pose to the livelihoods of the fishers have likely motivated them to invest in efforts to avoid, minimize, or mitigate their impacts on the broader marine environment. Furthermore, the HIMI fishery is a high-valued resource that is currently exploited by a low number of users (two companies) that possess secure and long-term rights to the resource. Small group size is generally thought to facilitate efforts to negotiate and implement agreements by reducing transaction costs (Olson, 1965); while secure property rights and the economic value of toothfish provide incentives to invest in the long-term sustainability of a resource (Ostrom, 1990; Grafton et al., 2006). These factors are clearly highlighted as critical by the fishers themselves:

> *"It [managing the MPA] involves lots of collaboration and cooperation between all parties, and a good understanding of the goals and attributes of MPAs. One powerful benefit available in the HIMI fishery, but not available in (for example) high seas fisheries, is the granting of secure, long term, fishing access rights. That also has a considerable impact on helping to focus on the longer term benefits of conservation and protection, as opposed to being constantly worried about 'will I have access next year' which, clearly, engenders a more short-term response and approach"*
>
> *(R. Arangio, Austral Fisheries, 29 June 2016).*

In other remote protected regions, meanwhile, such as the newly adopted Ross Sea region MPA in Antarctica, fisheries are competitive in that all fishing vessels lack individually assigned quotas and instead race to fish until the total quota is captured (Reid, 2019). While the Ross Sea region MPA planning process did include some fishing industry stakeholders, and the adopted MPA does accommodate commercial fisheries, it is unclear if the fishing industry will have similar incentives to participate in research and monitoring as at HIMI. Up to 16 different fishing companies from nine different fishing nation states compete for fisheries resources in the Ross Sea (CCAMLR, 2019). The race to fish is often cited as a core driver of overexploitation in fisheries, with corresponding impacts on the environment and the people that depend upon them (Grafton et al., 2006; Branch, 2009). As a result, it is unclear if participation alone will be sufficient to achieve similar results in the Ross Sea.

Second, although the remote nature of the HIMI Marine Reserve has certainly contributed to its success by limiting direct human interactions with the environment; human impacts on the prevailing climate regime are a growing threat to the HIMI Marine Reserve and the species it protects (IUCN, 2017; Whinam and Shaw, 2018). Because of the islands' location in the subantarctic region, occurring within the path of major circumpolar fronts, both the land and sea systems are highly vulnerable to climate change. King penguins and other species on HIMI, for instance, have depended upon foraging grounds located along these fronts (e.g., Peron et al., 2012; Bost et al., 2013). The environmental monitoring system supported by toothfish fishers and tourists which provides merely *ad hoc* monitoring of many species (with the notable exception of monitoring of toothfish and benthic surveys), may be insufficient to detect and respond appropriately to emerging threats from climate change. As a result, although the efforts of the toothfish fishers are to be commended, further support from government stakeholders for scientific surveys on land and in the sea may be necessary to ensure the long-term sustainability of the HIMI ecosystem.

CONCLUSION

The global push for large MPAs have led to an increasing number of relatively vast and remote protected areas that pose significant management, research, and monitoring challenges. Here we presented a unique case of the HIMI Marine Reserve – one of the most remote MPAs on earth and relatively large at 65,000 km2 (since expanded to 71,000 km2) – and the collaborative management between the Australian Government and fishing industry in meeting the objectives of the Reserve. The Reserve has generally been successful at both supporting sustainable fisheries while also conserving biodiversity. Importantly, the Reserve has in part met its goals through being remote and isolated; little to no humans regularly visit the HIMI region besides commercial fishers. The fishers are prohibited from fishing in the Reserve and demonstrate high compliance, as a result of several factors – their involvement with zoning of the MPA, their desire to keep their exclusive quotas for lucrative toothfish, as well as both companies striving to be good corporate citizens (for example Austral Fisheries are, to date, the only certified carbon neutral fishing business in the world). The early involvement of the fishing industry in the MPA process facilitated continued collaboration throughout management; the industry invests in research and monitoring to support the objectives of the Reserve while also aiding in monitoring and reporting any illegal fishing activities. However, mainly due to lack of capacity by the Australian Government, research, management and enforcement is largely passive with very little information on the status of species and ecosystems around HIMI. Given the future threat of climate change, current management may be insufficient at conserving the HIMI marine ecosystem. Additional support is needed from government, scientists and other stakeholders. Further, while this model works relatively well at HIMI, it may not apply to other remote MPAs. Only two companies fish in the HIMI EEZ and they have exclusive quota rights. In contrast in other remote

MPAs, e.g. the Ross Sea, where more than a dozen companies compete to fish in 'Olympic-style' fisheries, all vessels involved compete for the available catch. Further, while the collaborative management between fishers and the Australian government has arguably been a success, it may not be enough to manage for future environmental change, invasive species or other threats.

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the manuscript and online at https://sesmad.dartmouth.edu/ses_cases/18.

ETHICS STATEMENT

The committee is the University of Victoria Human Ethics Research Board. Procedure: We obtained informed consent from all participants. As interviews were done remotely via telephone and/or through email exchanges, we obtained verbal informed consent during interviews, and written consent in email exchanges. All participants reviewed the manuscript and approved the use of their names, organizations, and quotes where relevant.

AUTHOR CONTRIBUTIONS

CB, GE, and NB designed the research. CB and GE carried out the research, including interviews, and conducted the analyses. All authors wrote the manuscript.

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frontiers in Marine Science

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Progress on Implementing Ecosystem-Based Fisheries Management in the United States Through the Use of Ecosystem Models and Analysis

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Worldwide fisheries management has been undergoing a paradigm shift from a single-species approach to ecosystem approaches. In the United States, NOAA has adopted a policy statement and Road Map to guide the development and implementation of ecosystem-based fisheries management (EBFM). NOAA's EBFM policy supports addressing the ecosystem interconnections to help maintain resilient and productive ecosystems, even as they respond to climate, habitat, ecological, and social and economic changes. Managing natural marine resources while taking into account their interactions with their environment and our human interactions with our resources and environment requires the support of ecosystem science, modeling, and analysis. Implementing EBFM will require using existing mandates and approaches that fit regional management structures and cultures. The primary mandate for managing marine fisheries in the United States is the Magnuson-Stevens Fishery Conservation and Management Act. Many tenets of the Act align well with the EBFM policy, however, incorporating ecosystem analysis and models into fisheries management processes has faced procedural challenges in many jurisdictions. In this paper, we review example cases where scientists have had success in using ecosystem analysis and modeling to inform management priorities, and identify practices that help bring new ecosystem science information into existing policy processes. A key to these successes is regular communication and collaborative discourse among modelers, stakeholders, and resource managers to tailor models and ensure they addressed the management needs as directly as possible.

Keywords: ecosystem-based fisheries management, ecosystem modeling, fisheries science, fisheries management, natural resource management

INTRODUCTION

Worldwide thinking on fisheries management priorities has been moving away from the mid-20th century paradigm of fishing down our fish stocks with the expectation that we can achieve maximum sustainable yield from all stocks in all ecosystems simultaneously (Larkin, 1996; Link, 2018). The United Nations Convention on Biological Diversity, and the Food and Agriculture Organization (FAO), have created opportunities and principles for nations to individually and cooperatively develop ecosystem approaches to natural resource management, including for fisheries management Prins and Henne, 1998; Garcia et al., 2003; UN Fisheries, and Agriculture Organization [UNFAO], 2003; Un Fisheries, and Agriculture Organization [UNFAO], 2009). The FAO defines the ecosystem approach to fisheries as striving "to balance diverse societal objectives, by taking into account the knowledge and uncertainties about biotic, abiotic and human components of ecosystems and their interactions and applying an integrated approach to fisheries within ecologically meaningful boundaries" (UN Fisheries, and Agriculture Organization [UNFAO], 2003).

The National Oceanic and Atmospheric Administration (NOAA), the United States federal agency responsible for marine fisheries management, adopted an ecosystem-based fisheries management (EBFM) Policy and Road Map in 2016 to articulate the agency's goals for EBFM and practical steps to implementing those goals (NOAA, 2016). The EBFM Policy defines EBFM similarly to the FAO's definition for the ecosystem approach to fisheries: "a systematic approach to fisheries management in a geographically specified area that contributes to the resilience and sustainability of the ecosystem; recognizes the physical, biological, economic, and social interactions among the affected fishery-related components of the ecosystem, including humans; and seeks to optimize benefits among a diverse set of societal goals" (NOAA, 2016). Regardless of the particular definition of an ecosystem approach to fisheries or EBFM, managing natural marine resources while taking into account their interactions with their environment and our human interactions with our resources and environment requires the support of ecosystem science.

In the United States, our formal shift toward EBFM began in 1996 with amendments to the nation's marine fisheries management law, the Magnuson-Stevens Fishery Conservation and Management Act [MSA], 2010. Revisions to our national fisheries science and management priorities included: prohibiting overfishing and recovering overfished stocks, protecting essential fish habitat requiring minimizing bycatch, monitoring and managing the fishing gears permitted for use in marine waters, and a public planning process on exploring and expanding the application of ecosystem principles in fishery conservation and management [Pub. L. 104-297 (1996), Ecosystem Principles Advisory Panel [EPAP], 1998; MacPherson, 2001; deReynier, 2014]. Other United States laws that intersect with the MSA, and that affect the policy processes where ecosystem models may be beneficial include the Endangered Species Act (1973), which guides the recovery of threatened and endangered species; the Marine Mammal Protection Act of (1972), which prohibits the directed take and requires minimizing the incidental take of marine mammals; the Coastal Zone Management Act (1972), which coordinates coastal zone planning between U.S. states, territories, and the federal government, and the Coral Reef Conservation Act (2000). This broad array of federal laws, which still does not include all of the federal laws that address marine resources, intersect with similarly intricate laws from smaller regional jurisdictions within the United States (Crowder et al., 2006). Confusion among scientists about how best to interact with the policy processes associated with these laws is understandable and is likely a factor in the slow progress toward using ecosystem science to guide major policy progress beyond context-setting or improvements to individual species management.

This paper reviews example cases where scientists have had success in using ecosystem analysis and modeling to inform management priorities and stakeholder activities, and identifies practices that help bring new ecosystem science information into existing policy processes. We define successful use of ecosystem models in management processes in two ways: (1) management process success, such as the first time use of ecosystem modeling in a management process, where that modeling helped managers gain insights into interactions within their ecosystems; (2) resource outcome success, where the use of ecosystem models in a management process is expected to improve the health or status of particular fish stocks or habitats. Ecosystem modeling is relatively new to United States fisheries management processes and changes within natural systems are often difficult to monitor and detect; therefore, it may be some years before we can fully assess the success of our work as it may affect the overall health of marine ecosystems. While the case studies presented here are examples from United States marine resource management, the policy issues considered share priorities with fisheries conservation and management practices worldwide. Two of the case studies address management challenges related to setting fisheries harvest levels in changing ecosystems. Two additional case studies address estuarine and marine habitat conservation, and the final case study concerns bycatch minimization.

CASE STUDIES OF ECOSYSTEM MODELS IN PRACTICE

The United States has been using the integrated ecosystem assessment (IEA) framework (Levin et al., 2008, 2009) to develop collaborative scientific assessment and policy planning for managing marine resources and habitats. Both the IEA framework and the EBFM Policy include steps toward achieving EBFM that emphasize collaboration and consultation between scientists, policy-makers, stakeholders, and the public. Among the most critical research tools in the United States EBFM effort have been ecosystem models, which can assimilate diverse streams of information and support simulation tests of retrospective or future scenarios, scaled to the ecosystem or management issue in question (Latour et al., 2003; Pikitch et al., 2004; Townsend et al., 2008; Espinoza-Tenorio et al., 2012). Using ecosystem models in support of natural resource management requires not only the careful consideration and analysis of key ecosystem interactions, but also an understanding of where and how policy processes provide opportunities for considering the outputs of those models. Each large marine ecosystem, each nation, and regional governing bodies within nations, will have varying policy processes with varying needs and opportunities for using ecosystem models. Recognizing and working within the practical constraints of those policy processes will improve the use and uptake of ecosystem models in natural resource management (Cormier et al., 2017).

While the term "ecosystem model" has specific meaning in some marine science disciplines, we define the term as a wide range of modeling and analysis tools that are used to support the implementation of EBFM. These tools include conceptual models and related analytical approaches (Harvey et al., 2016) and a variety of biophysical, multispecies, food-web and end-to-end ecosystem models (further described in Plagányi, 2007; Townsend et al., 2008). This range covers models and analysis that consider only a few external factors influencing a single fish stock to a more holistic set of factors (e.g., climate, currents, biogeochemistry, fisheries, human dimensions; Rose et al., 2010; Fulton et al., 2011) influencing multiple, interacting fish stocks. While ecosystem models vary in terms of complexity, software platforms, scale, and scope, ecosystem modelers often adopt similar approaches ("best practices") to developing models designed to address a marine ecosystem management issue (Townsend et al., 2008; Un Fisheries, and Agriculture Organization [UNFAO], 2008). For example, the five case studies presented herein are at different stages of development, but each case generally follows five steps: identifying the problem and related management process; conferring among scientists, managers and stakeholders; review of initial model results; incorporating additional information; and exploring management actions (**Figure 1**). The narratives below illustrate the flow of these steps in greater detail.

Atlantic Herring Management Strategy Evaluation

This Atlantic herring case study discusses a process (**Figure 2**) that integrated information and analyses for several species and fisheries occurring off the Northeastern United States. Participants in this process worked through the New England Fishery Management Council (NEFMC), one of eight United States regional fishery management councils authorized under the MSA to provide advice to the United States government on fisheries management and regulations for activities within the United States exclusive economic zone. This case study provides an example of a novel fisheries management question explored through a traditional policymaking process.

Forage fish are ecologically important links between lower trophic level production and economically and socially important top predators. In many ecosystems, there are also commercially important fisheries targeting forage fish. There has been considerable interest in balancing the direct harvest of forage fish (a provisioning service for humans) with the supporting ecosystem services that they provide (e.g., Cury et al., 2011; Pikitch et al., 2012; Essington et al., 2015; Hilborn et al., 2017). In 2016, the NEFMC initiated a management strategy evaluation (MSE) to develop a harvest control rule for Atlantic herring, *Clupea harengus*, that considered herring's ecological role as forage (Deroba et al., 2019; Feeney et al., 2019). The harvest control rule needed to meet all MSA requirements, in addition to considering herring's role as forage for commercially and recreationally important fishes and for protected predators such as seabirds and marine mammals (Overholtz and Link, 2007). Among other fisheries, herring harvests contribute to the success of the Maine lobster, *Homarus americanus,* fishery, which uses herring as bait (Ryan et al., 2010; NMFS, 2016).

To understand the interests of many and varied stakeholders with diverse and potentially conflicting objectives, the NEFMC implemented a transparent and inclusive process to select an ecosystem-based harvest control rule and analyze its ecological and economic effects. The MSE was bounded by open stakeholder workshops in recognition of both MSE best practice and MSA requirements for public processes (Feeney et al., 2019). MSE models and analyses were tailored to the specific objectives and performance metrics outlined at the first stakeholder workshop, and were constrained by the management timeline to provide results by early 2017, less than a year after the planning process began (Deroba et al., 2019). This constraint motivated the use of existing models or newly developed models that were relatively simple but still adherent to best practices for multispecies management (Plagányi et al., 2014; Collie et al., 2016; Punt et al., 2016b). Following Plagányi and Butterworth (2012), a previously developed Atlantic herring population model was linked to simple deterministic delay difference models for three predators: bluefin tuna *Thunnus thynnus*, spiny dogfish *Squalus acanthias*, and common tern *Sterna hirundo*.

The MSE process prioritized use of data specific to the Northeast United States continental shelf ecosystem (e.g., Overholtz et al., 2000; Overholtz and Link, 2007; Link et al., 2008; Logan et al., 2015). Multispecies model parameters also drew from regional studies, such as information on herring-bluefin tuna relationships (Golet et al., 2015). Stakeholders helped fill important gaps: some workshop participants worked at seabird refuges and contributed essential data on common tern colony size, fledgling production, and fledgling diet (Deroba et al., 2019; Feeney et al., 2019). In contrast, the relationship between herring and spiny dogfish had to be hypothesized based on trend analysis rather than a clear mechanism, and a lack of existing information prevented the development of delay-difference models for any marine mammal predators of herring (Deroba et al., 2019). Economic analyses were linked to herring population model outputs and limited to performance of the herring fishery (Deroba et al., 2019).

An initial outcome of the MSE process was that several classes of control rules that performed poorly for predators, herring, and the herring fishery were eliminated by consensus at the second stakeholder workshop in December 2016. This left thousands of potential control rules with acceptable predator performance to be further narrowed based on performance for the herring fishery and the herring stock. After many follow-up questions regarding the performance of individual

FIGURE 1 | Interactive Process for Developing and Using Ecosystem Models in Policymaking.

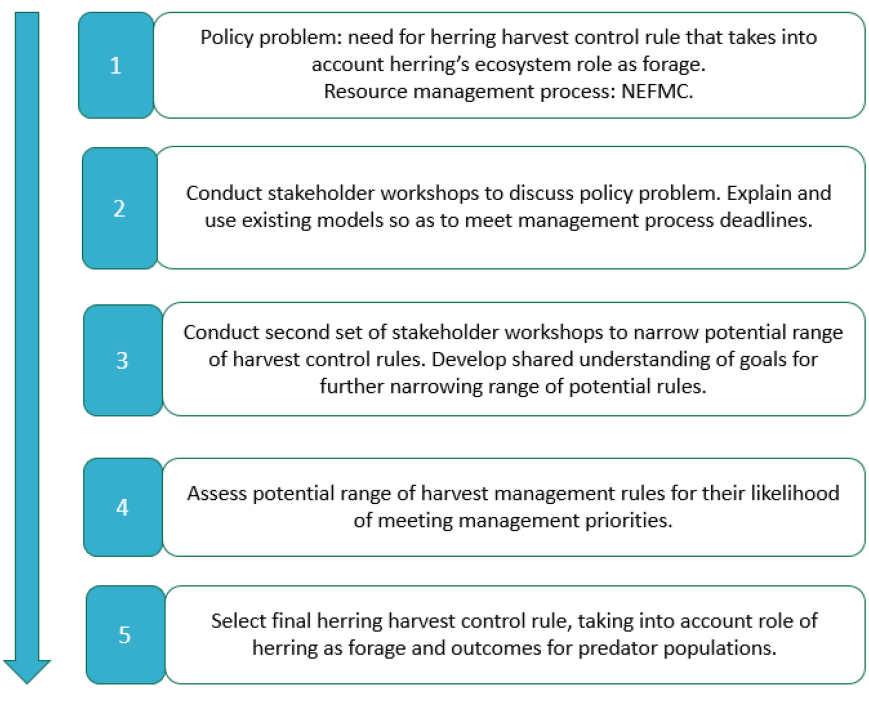

FIGURE 2 | Atlantic Herring Case Study Process.

control rules, the NEFMC narrowed the list to ten alternatives for further consideration and National Environmental Policy Act analysis (Feeney et al., 2019). Later, in September 2018, the NEFMC selected a final herring harvest control rule, which took into consideration the role of herring as forage and the outcomes for predator populations, setting aside a portion of the available catch to explicitly account for the important role of Atlantic herring as forage within the ecosystem (New England Fishery Management Council [NEFMC], 2018).

Conducting a multispecies MSE within an existing United States fishery management council process had no precedent, and therefore no formal structure. NEFMC members and staff, NOAA fisheries scientists and policy analysts shaped the MSE collaboratively to ensure that the stakeholder workshops and resulting analyses would be useful in NEFMC decision making. Communication between the managers, analysts and interested stakeholders took place throughout the process (Feeney et al., 2019). During the process, an external expert committee reviewed the MSE, endorsed it as best available science for NEFMC decision making, and suggested possible improvements for future iterations (Feeney et al., 2019). Although no schedule for revisiting control rule performance has been set, standard practice is to evaluate management procedures based on MSEs at approximately 5-year intervals (Plagányi et al., 2007; Rademeyer et al., 2007; Punt et al., 2016a). Other outcomes of this process include improved understanding of both the MSE process and multispecies interactions in the New England region (Feeney et al., 2019). The MSE process succeeded in introducing a wider range of ecological information into the larger fishery management council process, and supported strategic decision making based on simple multispecies modeling approaches. Overall, the NEFMC balanced multiple objectives in refining herring management, but it stated that it selected its control rule "to explicitly account for the important role of Atlantic herring as forage within the ecosystem" (New England Fishery Management Council [NEFMC], 2018). Considering ecological objectives is a critical first step toward the routine use of ecosystem analysis and modeling in fishery management. In the future, addressing societal benefits across a wider range of predators, ecological feedbacks, and fishery interactions would allow us to more fully evaluate harvest control rules.

Gulf of Alaska Pacific Cod Harvest

This Gulf of Alaska Pacific cod harvest case study (**Figure 3**) discusses how the North Pacific Fishery Management Council (NPFMC) addressed a surprising environmental shift that challenged their customary assessment and harvest rule setting process (**Figure 4**; Barbeaux et al., unpublished). Although the policymaking process is similar to that discussed in the Atlantic herring case study, the management question in this case study was driven by forces external to that process.

Alaska supports the largest federally managed fisheries in the United States, with landings of groundfish—such as Pacific cod *Gadus macrocephalus*, walleye pollock *Gadus chalcogrammus*, sablefish *Anoplopoma fimbria*, and various flatfish—that totaled 2.3 million metric tons in 2016 (Fissel et al., 2017). Under the MSA, the NPFMC establishes annual catch limits based on recommended allowable biological catch (ABC) from, in most cases, age-structured stock assessment models. NOAA's Alaska Fisheries Science Center supports EBFM by directly incorporating ecosystem-informed parameters into a few stock assessments (Marshall et al., 2018), and by presenting additional ecosystem information in tandem with individual stock assessments (Zador et al., 2017a). This coordinated process allows for ecosystem information, which is a synthesis of myriad data sources and model outputs, to be used to support any proposed reduction from the maximum ABC recommended by an individual stock assessment.

In 2017, the abundance estimates of Pacific cod, as observed in the biennial Gulf of Alaska bottom trawl survey of groundfish stocks, dropped by 71% from the previous survey in 2015 (Barbeaux et al., 2018). This resulted in an 80% reduction in the stock assessment-derived ABC from the previous year and a 77% reduction of what had been expected from previous assessments. The drastic change in the final catch limit that was set for 2018 represented a major impact on the Gulf of Alaska fisheries that target Pacific cod as well as towns such as Kodiak, Alaska, where Pacific cod play a vital role in the local economy in this rural, fishing-dependent island community (Himes-Cornell et al., 2013).

The Gulf of Alaska experienced an unprecedented marine heatwave from 2014 to 2016, which caused persistent and widespread sea surface temperature increases of 1–2°C and extensive ecological responses (Bond et al., 2015; Di Lorenzo and Mantua, 2016; Zador and Yasumiishi, 2017). The NPFMC had been informed of the changes in the ecosystem during the heatwave (**Figure 3**), but Pacific cod was the first managed stock to show a steep decline that could be explained in part due to the heatwave (Zador et al., 2017a; Barbeaux et al., 2018). Collaboration among the stock assessment author and ecosystem scientists resulted in: (1) an explanation of the Pacific cod decline due to heatwave-related increased adult mortality and lack of recruitment in the stock assessment (Barbeaux et al., 2018), and (2) an ecosystem model-based assessment of Pacific cod bioenergetics and diet limitations within the context of trophic-level wide negative impacts of the marine heatwave in the ecosystem assessment (Zador and Yasumiishi, 2017).

From the time when the trawl survey data were available through the setting of the final Pacific cod catch limit (∼3 months), ecosystem and stock assessment scientists, fisheries managers, and industry stakeholders communicated frequently about findings under development (Barbeaux et al., unpublished). Communication occurred among all three groups during formal processes such as management and industry meetings and informally through direct communication, with the end result that the final, drastic cut to the catch limit was accepted without controversy, demonstrating success in the management process. The NPFMC has a longer familiarity with ecosystem models and information than many of the fishery management councils; that familiarity ultimately supported their swift response to challenges for a particular fish stock within an ecosystem perturbed by climate variability. The ecosystem science-based explanations for the Pacific cod decline were integral to building trust among stakeholders in the

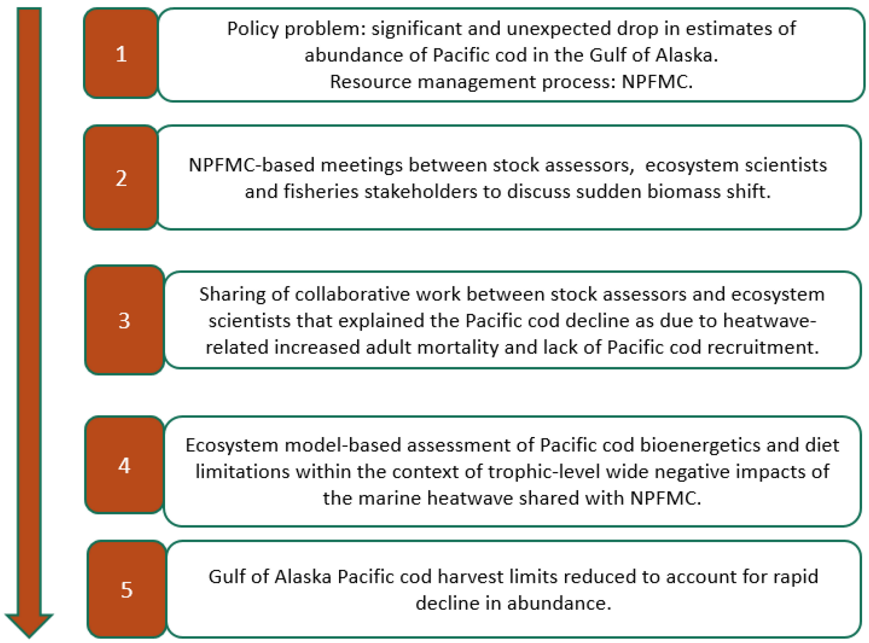

FIGURE 3 | Gulf of Alaska Pacific Cod Case Study Process.

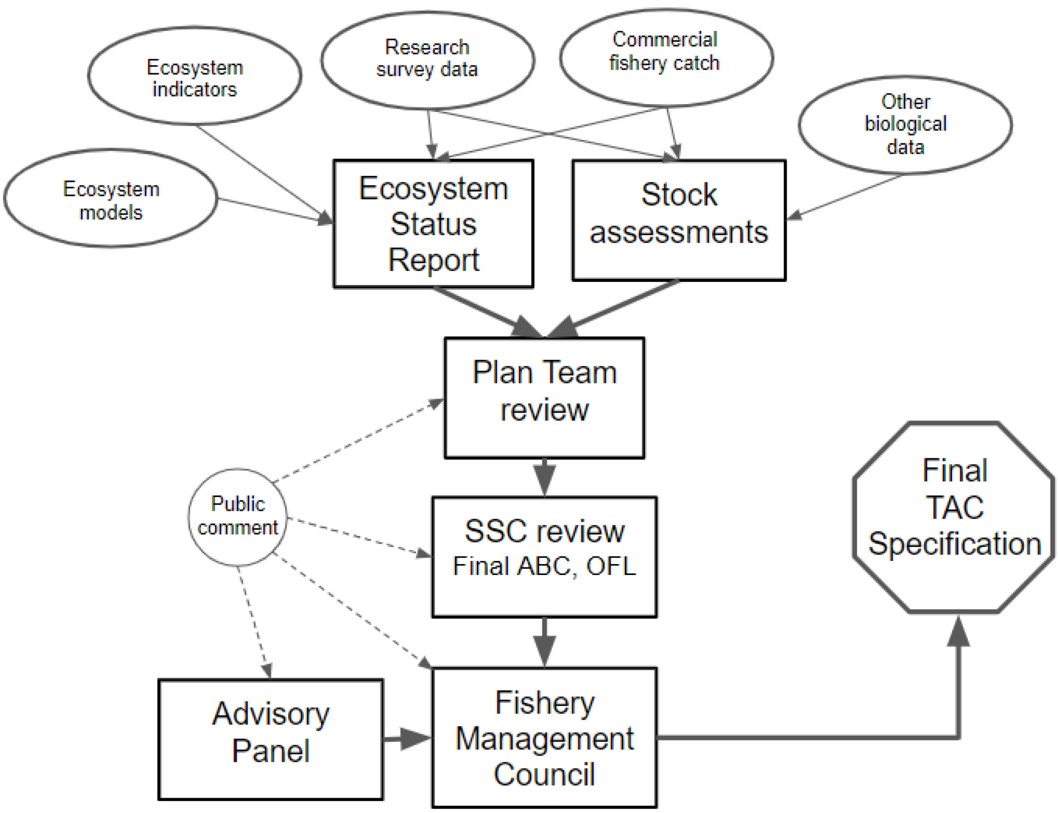

FIGURE 4 | Summary of annual groundfish federal management cycle in Alaska, from assessments through council review and final catch quotas. ABC, allowable biological catch; OFL, overfished level; TAC, total allowable catch (i.e., catch quota).

management decisions. Growing recognition of the importance of communication and transparency for successful EBFM has provided further impetus to formalize the incorporation of ecosystem science into fisheries management processes. This challenge is being met with the development of a suite of climate-informed ecosystem models, a fishery ecosystem plan that incorporates conceptual and quantitative models, and development of risk tables within stock assessments that include quantification of ecosystem concerns external to the assessment models (Barbeaux et al., unpublished).

Coastal Louisiana Restoration

For this case study, we move beyond the fishery management council process to habitat-based processes that support United States natural resource management priorities for estuaries and coastal zones (**Figure 5**). The MSA requires characterizing and protecting essential fish habitat, and the Coastal Zone Management Act (1972) requires the United States federal government to work with U.S. states and territories to support regional approaches to preserving and protecting the nation's coasts. A priority for fisheries ecosystem planning under the EBFM Road Map is: "Facilitating the participation of external federal, state (including territories), and tribal partners in the EBFM process by assessing the cumulative effects of human activities on marine ecosystems to help partners minimize the effects of non-fishing activities on trust living marine resources and habitats." This coastal Louisiana restoration case study explores the use of ecosystem models in an emerging and novel cross-mandate policymaking process.

The U.S. state of Louisiana, with much of its ecology shaped by the Mississippi River and its coastline on the Gulf of Mexico, is experiencing substantial losses of coastal land due to channelization of the outflowing Mississippi River and due to land subsidence. Multiple federal and state authorities have interests in and mandates related to freshwater marshes and coastal habitat restoration and protection in the Mississippi River Delta. Louisiana's Coastal Protection and Restoration Authority (CPRA) serves as a central authority for the state's coastal management entities. CPRA coordinates state coastal habitat restoration and cooperation with federal authorities on hurricane response. CPRA's Louisiana Coastal Master Plan is intended in part to guide the construction of large-scale sediment diversion projects to partially redirect the flow of the Mississippi River and provide sediments to rebuild depleted marsh habitat (Coastal Protection and Restoration Authority of Louisiana [CPRA], 2017). Ecosystem models have been essential for linking together the complex dynamics and currencies that span the interdependent terrestrial, aquatic, marine and social-economic components of this system.

Federal agencies working on and with sediment diversion projects in Louisiana have interests in the engineering aspects of these projects under the United States Army Corps of Engineers (USACE), in natural resource management aspects under the United States Fish and Wildlife Service and NOAA, and in hazardous weather monitoring and response under NOAA. Together, these agencies developed a USACE feasibility study on the potential effects of a suite of sediment diversion projects, and a NOAA plan for ecosystem science and modeling of the proposed diversion projects. To develop its feasibility study, the USACE formed a project development team of federal and state agencies with habitat and fish and wildlife authorities in the region (U.S. Army Corps of Engineers [USACE], 2012). Scientific work in support of the team crossed disciplines to evaluate the potential effectiveness of a suite of diversion projects at rebuilding marshes and the potential consequences of habitat changes on living marine resources and the coastal and marine ecosystem.

Coastal Protection and Restoration Authority and USACE used hydrological and ecological models to coordinate the analysis and planning process for sediment diversion projects. The Delft-3D hydrological model was used to predict the flow of sediment, salt, and other water quality components with and without the operation of the proposed diversion projects (Meselhe et al., 2015). The Delft-3D model allowed agencies to evaluate the effectiveness of diversions for rebuilding marsh habitat, and the influence on water quality factors like salinity. Outputs and ancillary products from the Delft-3D model were used as inputs for two ecosystem models, Ecopath with Ecosim and Ecospace, and the Comprehensive Aquatic Systems Model (Expert Panel, 2014; de Mutsert et al., 2017). These ecosystem models were used to evaluate the effect of diversion operations on the biomass and distribution of key fishery species.

Modelers for the hydrological and ecological models regularly consulted with the multi-agency project development team during the model development processes. These interactions with agency stakeholders helped modelers to understand expectations for the analysis and to get access to needed data. Conversely, agency stakeholders were able to understand the capabilities and limitations of the models and analysis. One key limitation for the ecological models was the lack of long time series of data on important ecological groups, a common challenge in ecological modeling (de Mutsert et al., 2017).

This integrated policy-making scientific modeling process was the first of its kind for the ecosystem-scale projects proposed for the region. Development and review of models was somewhat *ad hoc*; although the process was successful enough to serve as a framework for planned future analyses of two of the sediment diversion projects (e.g., the Mid-Barataria and Mid-Breton sediment diversion projects). CPRA commissioned an expert panel to provide independent review of the models and analyses, and presented results to the agency stakeholders.

The initial multiple-model evaluation of all of the sediment diversion projects in the region allowed CPRA to proceed with a sediment diversion project in the Barataria Basin of Louisiana (Coastal Protection and Restoration Authority of Louisiana [CPRA], 2014). Stakeholders in the Barataria Bay area have asked for more precise estimates of how the diversion operations would affect the biomass and distribution of key living marine resources as well as the ecosystem structure. Modelers on the project have emphasized the difficulty of making long-term projections about complex ecosystems. To account for and adapt to these concerns, scientists and natural resource managers in the region are investigating adaptive management approaches to ensure that system monitoring and modeling is ready for the eventual implementation of sediment diversion projects.

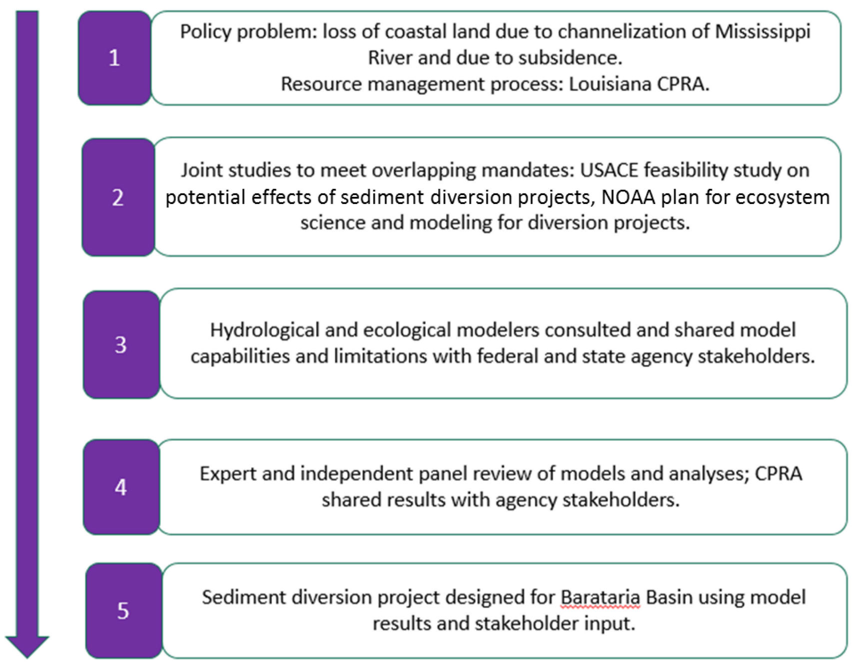

FIGURE 5 | Coastal Louisiana Restoration Case Study Process.

Ecosystem models are being used at the next stage of restoration planning for a diversion in Barataria Bay that will characterize food web dynamics under current conditions. In addition, these models are being used to guide development of a monitoring and adaptive management plan for the restoration process. It is too soon to evaluate resource outcome success for this process because management actions are still under review and have not yet been implemented.

Hawai'i Coral Reefs

Like the Louisiana case study, this Hawai'i case study focuses on marine habitat (**Figure 6**). In the United States, coral reefs are protected under a variety of laws, including MSA and CZMA discussed above, as well as the Coral Reef Conservation Act (2000). This case study was initiated through the regional Integrated Ecosystem Assessment (IEA) process (PIFSC, 2016). By bringing together scientists, policy makers and an engaged community, and the overall desire to reverse the declines in both coral cover and fish biomass, there was an interest in exploring various management regulations that could mitigate or reverse the downward trend in natural resources. Additionally, in 2016, the governor of Hawai'i pledged to "effectively manage" 30% of the marine areas along Hawai'i's coastline by 2030. However, defining "effectively" is left up to the managers of the Hawai'i Division of Aquatic Resources (DAR) through a multi-year spatial planning exercise. This case study looks at using ecosystem models to consider issues that cross federal, state, and local mandates and processes.

In Hawai'i, coral reef ecosystems are degrading in many regions due to land-based pollution, fishing, coastal development and other local stressors combined with the devastating 2015 coral mortality from ocean warming (Friedlander et al., 2008; Couch et al., 2014; Bahr et al., 2017). Though coral reefs can recover over decades, climate models project that coral bleaching related mortality may occur annually within the next 20–25 years (van Hooidonk et al., 2016). Changes in marine resource management are needed to improve recovery of ecosystem structure and services. The majority of Hawai'i's reefs are within state waters, however, under the Coral Reef Conservation Act, and specified in the EBFM road map, NOAA works with jurisdictions to support coral reef conservation and management. Hawai'i's coral reef management embraces an ecosystem-based approach to management to guarantee that ecosystem services such as fishing and a resilient ecosystem structure are maintained or improved.

Two recent efforts to support local decision making have included the development of ecosystem models. In both cases, the local managers identified the management scenarios for model simulation. One effort was led by the University of Hawai'i at Mānoa (UH) and involved Pacific Islands Fisheries Science Center (PIFSC) scientists in developing the model, while the second effort resulted from PIFSC discussions through its IEA process. In both cases, scientists proposed to develop a model, and alternative management strategies were identified in consultation with DAR. The ecosystem modeling platform used by UH was HIReefSim (Hawai'i Reef dynamics Simulator) for the islands of Maui, Moloka'I, and Lana'i (Weijerman et al., 2018b). HIReefSim details dynamics of five benthic groups (three algal and two coral groups) and two fish groups (herbivorous and piscivorous fish) and is based on gridded (500 × 500 m) base maps of initial conditions and main stressors, such as climate change (leading to coral mortality), land-based sources of pollution and fishing (Melbourne-Thomas et al., 2011). Selection of this model was based on its compatibility with the DAR effort of selecting areas (grid cells) using MARXAN (Ball et al., 2009) and the fact that HIReefSim can simulate land-sea dynamics. The idea being that results from MARXAN identified areas where management would be warranted and HIReefSim could evaluate the tradeoffs of alternative management options that include land based and marine based scenarios. The modeling software used in the IEA effort was Ecopath with Ecosim (EwE; Polovina, 1984; Christensen and Walters, 2004) for Puako Bay on the West Coast of Hawai'i Island (Weijerman et al., 2018a). Selection of EwE was based on its focus on just one bay and the ability to include all fish species and gear restriction as management options.

The process of developing the model was *ad hoc*. Models were developed by PIFSC scientists in collaboration with scientists from UH, the United States federal Environmental Protection Agency, Gulf Ecology Division, Oregon State University for HIReefSim, and with staff from The Nature Conservancy and DAR for EwE. In both cases, the developed models were used as examples of how ecosystem models can be used as decision support tools in the face of climate change by quantifying socio-ecological tradeoffs of alternative fisheries and land-based management policies. UH is in constant dialog with DAR to discuss the usefulness of HIReefSim as a management-support tool for their spatial planning. The EwE model was developed in collaboration with DAR to ensure that the simulated policy regulations were relevant for actual implementation. Results showed that the "only line fishing" scenario in combination with a reduction in nutrients and sediments generated the most balanced trade-off between marine resource users and ecosystem resilience. In both cases, results were presented to DAR and were well received. Upon request from DAR, HIReefSim is now being parameterized for Kaneohe Bay on the windward coast of Oahu and there is also interest for developing HIReefSim for other areas. The results of the EwE model were presented at a regional IEA symposium to the public, scientists, non-governmental organizations, fishers and managers. DAR has since requested similar model development for other geographic areas, but potential further development awaits funding.

From a management process perspective, ecosystem modeling and analysis was used for strategic management decisions, i.e., to get insight in the socio-ecological tradeoffs in alternative marine and land-based management strategies. Interest in the use of ecosystem models to evaluate potential changes in ecosystem structure attributable to changes in water quality (e.g., temperature, nutrients) and fish biomass (e.g. herbivorous fishes) is present. Models output have highlighted potential impacts to various stakeholders (tourist, fishers) and the overall ecosystem structure and resilience. At this stage, resource outcome success cannot be evaluated as the DAR spatial planning exercise is still ongoing, and the state of Hawai'i has not yet decided what constitutes "effective" management. However, DAR's positive reception of the regional ecosystem models, provides a useful example of how to incorporate ecosystem models into the decision making process and will facilitate the ongoing discussion between UH, PIFSC and DAR about "effective" coastline management into the future.

Dynamic Ocean Management in the California Current Ecosystem

For this final case study, we return to the fishery management council process to look at bycatch minimization off the United States West Coast (**Figure 7**). Dynamic ocean management in the California Current Ecosystem combines multiple species distribution models to emulate a simple ecosystem model, providing nowcasts for potential bycatch issues in fisheries for highly migratory species. NOAA scientists worked with the West Coast Regional Office, Pacific Fishery Management Council (PFMC) and West Coast fishermen to make both the model and its inclusion into management processes and the fishery itself more practical for on-the-water use. Bycatch in these fisheries includes species protected under the Endangered Species Act and Marine Mammal Protection Act, laws with different and sometimes competing priorities from the MSA. The participating scientists, managers, and stakeholders had to use the fishery management council process to simultaneously meet these different priorities to come up with solutions that also worked for the fishing fleets subject to these laws.

Federal fishery management policies in the MSA are implemented through regional fishery management plans, which guide tactical decision-making for the management of individual stocks and species groups. Decisions such as harvest limits and allocations are often made annually, and decisions on where and when fishing can occur may range from short-term (seasonal) schedules to long-term designations of areas opened or closed to particular gear types. These scales and time frames for management are practical, given the time, effort and data required for conducting stock assessments, technical and public review, evaluating alternatives, and developing final decisions. In the United States, this process has been successful in maintaining sustainable fishing pressure on target species, rebuilding populations of overfished species, and reducing bycatch of protected species. However, the resulting plans can constrain opportunities for the fishing industry if the management action is conservative and overly lacking in flexibility for adjustments in space and time (Maxwell et al., 2015; Dunn et al., 2016). In addition, anomalous years can result in protected species shifting out of their normal habitats, where protections may be in place, and into unprotected waters, leading to crises such as mass entanglements of North Atlantic right whale (*Eubalaena glacialis*) in fishing gear during 2017 (Meyer-Gutbrod and Greene, 2018).

One solution is "dynamic ocean management," which employs species data and distribution models to provide fishermen with real-time spatial estimates or short-term forecasts of fishing conditions and bycatch risks (Hobday et al., 2014; Maxwell et al., 2015). Many dynamic ocean management tools employ species distribution models to track changes in ocean conditions and estimate the probabilities of encountering each target and bycatch species in a given area, which are then

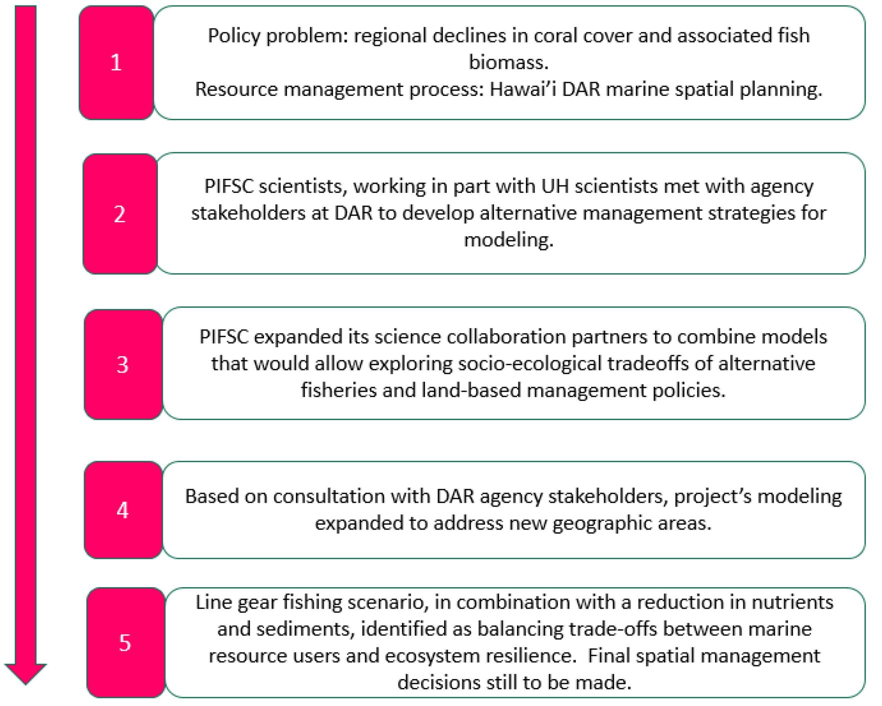

FIGURE 6 | Hawai'i Coral Reef Planning Case Study Process.

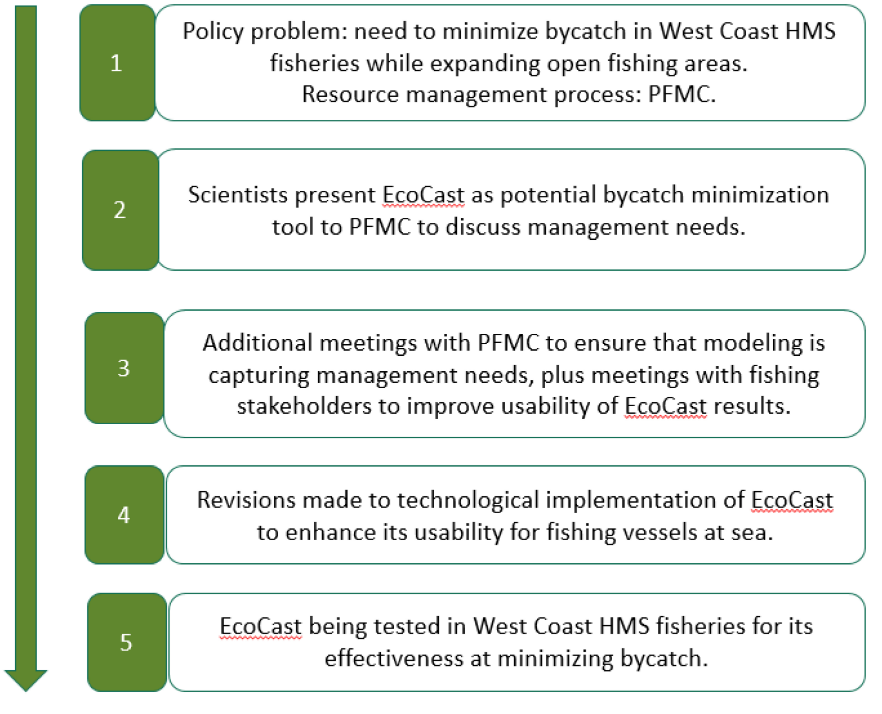

FIGURE 7 | California Current Dynamic Ocean Management Case Study Process.

combined using individual risk weightings to produce a single product (Hazen et al., 2018; Welch et al., 2019a). Providing output from these models thus allows managers and fishing vessels to assess fishing opportunities and risks of protected species bycatch at much finer spatiotemporal scales than large closures implemented over long time periods.

The EcoCast tool[1] is one example of a dynamic ocean management tool from the United States West Coast (Hazen et al., 2018). The California drift gillnet fishery targets swordfish (*Xiphias gladius*) and secondarily mako sharks (*Isurus oxyrinchus*) and thresher sharks (*Alopias vulpinus*) under the PFMC highly migratory species fishery management plan (Eguchi et al., 2017; Mason et al., 2019). Multiple management measures have coincided with the decline of fishery effort over the past 20 years (Mason et al., 2019). In particular, bycatch of protected species continues to constrain the fishery such that swordfish harvest in United States waters is well below maximum sustainable yield. In 2001, a drift gillnet fishery closure was implemented from August to November in a 552,000-km^2 area that encompassed 22 different bycatch events of federally endangered leatherback turtle (*Dermochelys coriacea*) (66 FR 44549, August 24, 2001; Eguchi et al., 2017). This closure severely limited drift gillnet fishing opportunity (Hazen et al., 2018; Mason et al., 2019). EcoCast was developed to provide a dynamic approach to test and improve on the static closed area, by providing nowcasts of target species catch (swordfish), protected species bycatch [leatherback turtle, California sea lion (*Zalophus californianus*)], and fish bycatch [blue shark (*Prionace glauca*)] (Hazen et al., 2018). In all scenarios, managers and fishers themselves face tradeoffs among catch and bycatch in where to fish; thus EcoCast implements a weighting scheme that reflects management priorities when coming up with estimates of catch and bycatch risk (Hazen et al., 2018). These estimates are produced daily, and additional analyses have been added to assess uncertainty caused by missing ocean data (e.g., poor satellite coverage; Welch et al., 2019a,b). At the request of fishermen and managers, EcoCast is being updated to use high-resolution ocean model output (daily and 10 km; Brodie et al., 2018) instead of remotely sensed data (daily and 25 km), and to incorporate new species such as protected cetaceans. Most recently, EcoCast was presented to swordfish fishermen with the hope of ultimately improving the utility of EcoCast with on the water validation.

The National Oceanic and Atmospheric Administration scientists first presented EcoCast to the PFMC in 2014 and 2015, after National Aeronautics and Space Administration had funded the project but before work had begun. This was done to alert the PFMC of the tool and to get feedback on the framework and approach. Draft models and data integration approaches were presented to PFMC advisory bodies for technical review in the fall of 2016. EcoCast was also presented twice to the fishing community during development to incorporate their feedback, and has been presented three more times to fishermen seeking fishing permits. As an example of the value of stakeholder feedback, one feature under development was a smart phone application for uploading opportunistic sightings data and to distribute model results, but many of the fishermen did not use smart phones, which lowered the value of the proposed feature. The EcoCast team thus developed two web-based alternatives available by smartphone or computer: a map product[2] that can be downloaded with the limited bandwidth available at sea; and an explorer tool[3] to explore risk weightings and how changing weightings affect the map product. The final product and tool was presented on the PFMC floor in November of 2017, going live at the same time for the 2017 fishing season. Researchers are currently working on exploring the utility of EcoCast for additional gear types, assessing the efficacy of the existing spatiotemporal closures, and identifying other fisheries where a similar dynamic modeling approach may be beneficial.

While EcoCast has been successfully explained to and shared with managers and the public, management of allowable fishing gears and locations for highly migratory species fisheries within the United States West Coast exclusive economic zone has been in flux for several years. EcoCast use will likely remain experimental in the near-term, although with benefits to fisheries and protected species. While this case study illustrates successful development of an ecosystem model and tools appropriate to fisheries management, it also shows that management processes often face challenges to using ecological models that are not at all associated with the quality or utility of the models themselves.

DISCUSSION

Each of these case studies provides an example of using some level of ecosystem modeling to advance EBFM. The Dynamic Ocean Management example used relatively simple analysis of simple spatial and gear interactions among species to improve options for minimizing bycatch in a particular fishery. The Coastal Louisiana and Hawai'i case studies used complex food-web models. The Atlantic Herring example used simplified food web interactions and economic models. The Gulf of Alaska case focused on environmental drivers and bioenergetic models.

In these examples, ecosystem modeling and analysis was used to support ecosystem-based fisheries management decision-making. The Alaska, Atlantic, and California examples, illustrate the use of ecosystem modeling in direct ecosystem-based fisheries management decisions. That is climate, habitat, ecological, or human dimensions information was quantified and used to adjust how at least one fishery was managed or to allow flexibility in how the fishery was executed. In the Hawai'i example, discussions on how to best implement EBFM to effectively manage 30% of the coastline are still on-going but these are informed by the modeling efforts. Although the Coastal Louisiana example was less focused on direct fishery management and more focused on habitat management, identifying and understanding essential fish habitat is an important component of the MSA's vision for fisheries management and habitats are a relevant unit of analysis to help operationalize EBFM (Marshak and Brown, 2017). For the habitat-oriented models (Hawai'i and Louisiana), a broader

[1] https://coastwatch.pfeg.noaa.gov/ecocast/

[2] https://coastwatch.pfeg.noaa.gov/ecocast/map_product.html

[3] https://coastwatch.pfeg.noaa.gov/ecocast/explorer.html

set of stakeholders and agencies were involved, and a less well-defined decision-making process (i.e., not a structured fisheries management council approach) was in place, so the efforts were more geared toward heuristic understanding of the focal systems and strategic management of habitats for supporting a broad set of fish stocks.

In these examples, participating scientists used best practices for model development and implementation. Over the past decade scientists and administrators at NOAA Fisheries Service have regularly convened a National Ecosystem Modeling Workshop (NEMoW) in support of living-marine resource management and to more formally review, evaluate, and support the ecosystem modeling efforts of NOAA Fisheries. The general objectives of the NEMoWs are: (1) to address broad questions of national interest for applied ecosystem modeling for living marine resource management, (2) to provide a forum for ecosystem modelers and scientists in the agency to network and share information on ecosystem modeling advancements and best practices, and (3) to provide a vehicle to advance ecosystem modeling within NOAA Fisheries as it meets its mandates and obligations. Scientists involved in these workshops have generated recommendations and best practices for ecosystem modeling and analysis and have documented them in a series of technical memoranda (Townsend et al., 2008, 2014, 2017; Link et al., 2010). The list of recommendations and best practices from these reports is extensive. A few major recommendations that have been discussed in multiple workshops include:

(1) Develop and maintain ecosystem modeling capacity and infrastructure, because anticipating management needs can help to ensure utility and relevance of modeling efforts.
(2) Apply iterative communication with managers and stakeholders; get them accustomed to seeing ecosystem models and analysis to build credibility.
(3) Ensure periodic review (informal: colleagues, stakeholders, managers) throughout model building, to avoid rejection of model at a late formal review stages when problems (model structure, mismatch of objectives, etc.) could be caught earlier.
(4) Use multiple models to address uncertainty in model structure and major ecosystem drivers - apply a range of multiple-model analytical or quantitative approaches appropriate for the type of question, the types of uncertainty, and amount of data available.
(5) Implement an MSE framework for providing and ecosystem context for living marine resource management and for exploring uncertainty.

Many of these best practices have been suggested elsewhere (e.g., Un Fisheries, and Agriculture Organization [UNFAO], 2008). Most of these best practices and recommendations have been implemented by ecosystem modelers and scientists in the United States, and as illustrated in these example cases, this has led to progress in advancing EBFM in the United States. A longstanding ecosystem modeling capacity and infrastructure for Alaska and the Northeast, enabled science centers to respond to requests by management councils to address their ecosystem concerns in a timely fashion. In those cases, prior regular, iterative communications between councils and ecosystem scientists had helped the scientists to build credibility in their analytical products and facilitated the use and further development of products for EBFM decision-making. In all of these examples, periodic informal review was used to help guide tool development and ensure a level of utility to managers and stakeholders. In the Dynamic Ocean Management case, regular stakeholder review was especially important for honing the tools and ensuring their utility. In the Coastal Louisiana and Hawai'i examples, a multiple model approach was implemented to allow exploration of the influence of drivers and stressors on focal ecosystems. This laid the groundwork for future ecosystem modeling and analysis to inform management actions. The Atlantic Herring case illustrates the use of MSE as an approach for providing ecosystem context to fisheries management and enabling managers to make an ecosystem-based decision.

Collaborative Processes for Ecosystem Model Use in EBFM

At the heart of each case study is dedicated collaboration between scientists, managers, and stakeholders from early in the process of ecosystem model development and use. As many authors have concluded, long-term commitment to collaboration is essential to successful EBFM because it clarifies objectives, promotes participation and information exchange, facilitates identification of tradeoffs, and ultimately builds trust and investment among parties (Peterman, 2004; Caddy and Seijo, 2005; Levin et al., 2009; Link et al., 2012). The collaborative process generalized in **Figure 1** and customized in different case studies provides real-world illustration of the efforts involved and the benefits derived.

The first step of the common collaborative framework, identifying the policy problem through an existing resource management process (**Figure 1**), is an essential starting point that has often proven difficult to achieve, perhaps because the initial question scientists ask ("What are your ecosystem objectives?") can be very abstract to managers and stakeholders in the absence of more specific context. Contemporary fisheries management has many very specific objectives articulated in legislation such as MSA and further honed in regional fishery management plans. Ecosystem objectives are not spelled out as clearly in legislation, and fishery management councils have only recently begun to adopt fishery ecosystem plans (FEPs) (Pew Charitable Trust Issue Brief, 2015). However, the case studies here are founded in specific policy questions that are anchored in existing resource management processes in which the managers and stakeholders are deeply invested (**Figure 1**). This is likely a better starting point from which to build and tailor ecosystem models than "what are your ecosystem objectives?"

The next two steps in **Figure 1** involve conferring early with managers and stakeholders to ensure that scientists have an appropriate understanding of the policy problem and the capabilities of different ecosystem models and products; and to frequently reconvene to ensure that all parties continue to correctly understand and characterize management needs and model results. Specifying and reaffirming such details is of great importance for several reasons. First, defining key details up front (e.g., appropriate spatial scales, temporal scales, currencies of model outputs, platforms of product delivery) improves the likelihood that model products will meet the expectations of managers and stakeholders. This should support more efficient implementation, and reduce the need for time-consuming, expensive revisions of models late in the process. Second, the initial dialog puts an emphasis on building models that are tailored to the specific management problem, rather than trying to redefine the problem so that it can be tested in an existing model with no major adjustments to the model structure. Tailoring the model to the problem is an essential best practice of ecosystem modeling). Third, dialog enables managers and stakeholders to become more familiar with the assumptions, strengths and weaknesses of a modeling framework. This transparency increases trust among participants; it also encourages managers and stakeholders to contribute their wealth of accumulated knowledge, which can help to identify specific mechanisms that a model should include, or to highlight information that can assist in parameter development or filling of gaps (Miller et al., 2010). The MSE workshop process described in the Atlantic herring case study (Deroba et al., 2019; Feeney et al., 2019) is an excellent illustration of how this emphasis on ongoing dialog reaps benefits throughout the course of ecosystem model development and implementation.

The fourth step is to assess additional scientific information and analyses needed to address the problem. This step emphasizes the need for the process to maintain sufficient flexibility to adapt as understanding of a management problem evolves or the nature of the problem changes. Several of the case studies above arose from events that required an urgency in response (e.g., rapid coastal erosion in Louisiana; sudden decline in Gulf of Alaska Pacific cod; bycatch of endangered leatherback turtles), and such events tend to incite considerable attention and research effort, which can generate important new information that can hone ecosystem-level management objectives and add valuable functionality to ecosystem models. This step can also further build trust and transparency among scientists, managers and the public, such as in the Atlantic herring case where stakeholders were able to provide data that filled modeling gaps. This step accentuates the value of having relatively simple, readily adaptable models as part of a model ensemble (e.g., Plagányi et al., 2014; Collie et al., 2016; Punt et al., 2016a). One additional example of a relatively simple, adaptable approach is qualitative network modeling, for instance used in the Gulf of Alaska (Zador et al., 2017b) and in the California Current IEA (Harvey et al., 2016). In these applications, qualitative modeling has proven useful for incorporating stakeholder and management input, understanding linkages from ecology to human communities, and modeling portions of the ecosystem that are not well sampled (i.e., "data-poor"). This step of addressing scientific analyses and data needs should also be valuable if one assumes that the problem is going to require long-term, adaptive management: building in a process for onboarding new information, from researchers, stakeholders, and local ecological knowledge (Ainsworth, 2011; Beaudreau and Levin, 2014) into ecosystem models should improve the value and efficiency of model-based decision support.

The final step in **Figure 1** is to explore the possible management actions that can lead to better, even if not perfect, outcomes under existing legal mandates and policy processes. This step is at the heart of MSE, as outlined in the Atlantic herring case study, and is also evident in the outcomes of the Gulf of Alaska cod and Louisiana sediment diversion case studies; the other two case studies have not reached the stage of full management implementation yet but the collaborative relationships are in place. This step further ensures a participatory process, in that the possible management actions and outcomes are best defined by the managers who will implement the actions and the stakeholders who will be affected by them.

CONCLUSION

This manuscript was intended as a broad, brief review to illustrate the use of ecosystem modeling and analysis to advance EBFM in the United States. A more focused review on specific aspects of modeling and EBFM could provide more detailed, actionable steps for making progress. This review highlights the processes and broad steps needed for continued progress toward EBFM and provides some evidence that best practices, when implemented, provide positive results.

To implement EBFM, a broad suite of tools beyond single species/stock population dynamics models are needed. These examples emphasize the need for models that incorporate a range of biophysical factors influencing stocks and interactions among fish stocks. They also illustrate a need to incorporate social and ecological factors. Most significantly, these case studies emphasize the need for ecosystem modelers to have or find access to policy-making processes. Where clear, established processes exist, like fishery management council processes, ecosystem modelers are introducing new science tools to those processes. Where policy-making processes have to be first designed by participants, modelers may face practical challenges in figuring out who to consult with and how their work might influence decisions.

A broad view of fisheries management is also necessary. Within the traditional single species/stock management process, there are limited controls (harvest regulations) to influence the status of a stock. However, other mandates enable some additional regulation of other external factors that influence stocks (e.g., habitat regulations). Moreover, the application of simple ecosystem models enables fisheries management (e.g., managing bycatch) on a finer time scale and with more agility than the conventional fisheries management processes. Ecosystem modeling can help make the connections between these external factors, their control, and stocks.

Communication among modelers is needed. Modelers focused on a particular ecosystem can benefit from regular interactions with modelers in other systems. Regular organized communication among model groups facilitates development, revision, and implementation of methodologies as well as developing and implementing best practices of model use.

Communication between modelers, managers, and stakeholders is essential. These examples demonstrated that early and frequent communication among these groups expedited model implementation and management decision-making. The cases highlighted that continued communication with stakeholders enabled model refinement and ensured their utility. While this level of communication has been beneficial, further iterative and systematic communication would also benefit scientists, managers, and stakeholders. Doing so would move modelers responding to urgent and critical events to providing strategic and tactical advice for holistic planning and operational use. One forum for this type of communication in the fisheries realm is annual Ecosystem Status Reports delivered to United States fishery managers through the IEA process (Zador and Yasumiishi, 2017; Gaichas et al., 2019; Harvey et al., 2019). Repeated communication allows modelers to transition from responding to urgent and critical events, to providing strategic advice for operational use. This can include holistic risk assessments (Gaichas et al., 2018; Samhouri et al., 2019) and planning for climate change effects, for instance for the California drift gillnet fishery considered by EcoCast (Smith et al., unpublished). This type of risk assessment can lead to further ecosystem modeling and analysis. Risk assessment is the first step in Mid-Atlantic Fishery Management Council's ecosystem process that also includes social-ecological conceptual modeling, potentially leading to MSE addressing social-ecosystem linkages (Gaichas et al., 2016).

Science in support of EBFM has evolved and grown over time, and relies on multidisciplinary models to illustrate and weigh the tradeoffs we make in managing natural marine resources. Ecosystem models can inform the IEA process and support EBFM efforts by: (1) synthesizing available data to help us understand and assess system dynamics, (2) testing scenarios of the risk to key species of top-down or bottom-up mediated stressors, (3) testing scenarios of the effectiveness and tradeoffs of management strategy alternatives at meeting the various requirements of a nation's or region's natural resource management laws and policies. Ecosystem modelers themselves can support EBFM efforts by reaching out to managers and stakeholders for a better understanding of management process challenges and possibilities.

AUTHOR CONTRIBUTIONS

All authors listed have made a substantial, direct and intellectual contribution to the work, and approved it for publication.

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Bridging From Monitoring to Solutions-Based Thinking: Lessons From CalCOFI for Understanding and Adapting to Marine Climate Change Impacts

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Multidisciplinary, integrated ocean observing programs provide critical data for monitoring the effects of climate change on marine ecosystems. California Cooperative Oceanic Fisheries Investigations (CalCOFI) samples along the US West Coast and is one of the world's longest-running and most comprehensive time series, with hydrographic and biological data collected since 1949. The pairing of ecological and physical measurements across this long time series informs our understanding of how the California Current marine ecosystem responds to climate variability. By providing a baseline to monitor change, the CalCOFI time series serves as a Keeling Curve for the California Current. However, challenges remain in connecting the data collected from long-term monitoring programs with the needs of stakeholders concerned with climate change adaptation (i.e., resource managers, policy makers, and the public), including for the fisheries and aquaculture sectors. We use the CalCOFI program as a case study to ask: how can long-term ocean observing programs inform ecosystem based management efforts and create data flows that meet the needs of stakeholders working on climate change adaptation? Addressing this question and identifying solutions requires working across sectors and recognizing stakeholder needs. Lessons learned from CalCOFI can inform other regional monitoring programs around the world, including those done at a smaller scale in developing countries.

Keywords: climate change, CalCOFI, fisheries, ecosystem management, ocean observing, monitoring

INTRODUCTION

The US West Coast is one of the best studied marine regions of the world due to a high concentration of academic institutions, investment in fisheries, and political support. Consequently, the data-rich environment provides an excellent opportunity to explore how oceanographic monitoring programs can support ecosystem management strategies that are adaptive to climate change. Ideally, more comprehensive oceanographic monitoring should lead to more climate-resilient management (Skern-Mauritzen et al., 2016; Davidson et al., 2019), however, there are unique challenges in linking physical and biological oceanographic data to management needs (Field and Francis, 2006). These include, for example, difficulties in linking meaningful climate indices to fisheries trends to inform management decisions (Myers, 1998; Smith et al., 2007; Pitcher et al., 2009), and in creating data products that support the needs of a diverse community of stakeholders (Rayner et al., 2019).

One reason the US West Coast is so well studied is due to the long-running California Cooperative Oceanic Fisheries Investigations (CalCOFI) Program. CalCOFI began in the late 1940s in response to the collapse of the economically important sardine fishery on the US West Coast (Hewitt, 1988; Scheiber, 1990). Competing hypotheses argued for overfishing versus environmental change as the dominant driver responsible for the sardine decline, and the CalCOFI Program was founded to investigate the relationship between oceanographic conditions and sardine abundance (Hewitt, 1988). From the beginning, CalCOFI represented a unique partnership between academic, federal, and state partners (Scheiber, 1990), which now consists of the Scripps Institution of Oceanography at UCSD, federal scientists at NOAA's National Marine Fisheries Service (NMFS), and scientists working for the California Department of Fish and Wildlife (Ohman and Venrick, 2003; McClatchie, 2014). Since 2004, CalCOFI also contributes to the California Current Ecosystem Long Term Ecological Research (CCE LTER) Network (Ohman and Hobbie, 2008).

While initially focused on sardines, the scientists who founded CalCOFI had the vision to create an ecosystem monitoring program at a time when ecosystem research was in its infancy (Scheiber, 1997), thus beginning one of the most comprehensive ecosystem monitoring programs that extends the full length of the Keeling Curve (Keeling and Keeling, 2017) – a time series that is fundamental to the scientific understanding of increasing greenhouse gases and anthropogenic climate change. Presently, the overarching objective of CalCOFI is to understand the effects of long-term changes in the California Current Ecosystem (CCE) (Ohman and Venrick, 2003).

The goal of this study is to identify how the CalCOFI Program can best support stakeholder needs for understanding climate change impacts and developing climate change adaptation strategies. To do this, we first review the data that are collected by the CalCOFI Program (1949-current) and synthesize how these data have informed our understanding of the dynamics and climate sensitivity of the CCE. In doing so, we also examine the CalCOFI record for climate relevant variables and compare changes observed over the last 70 years to changes predicted by climate models for the future. We then consider how CalCOFI data are currently used to support ecosystem and fisheries management decision-making, explore these within the context of adaptation to climate change, and consider how CalCOFI data can be used by a diversity of stakeholders to support climate change adaptation needs in the future.

MATERIALS AND METHODS

What Data Does the CalCOFI Program Collect?

At present, CalCOFI conducts quarterly ship-based surveys in a grid design at fixed stations in the Southern California Current Ecosystem (SCCE) and collects both hydrographic and biological samples. The earliest samples were collected in 1949, and the sampling protocols were standardized by the early 1950s, allowing for comparisons across the 70 years time series. Although the survey region has expanded and contracted, the "core" sampling grid, consistently maintained throughout the time series, is composed of 66 stations along 6 grid lines (**Figure 1**). This area ranges from the US-Mexico Border to slightly north of Point Conception, California, and extends over 500 km offshore. In 2004, nine additional coastal stations were added to the "core" CalCOFI grid, as part of the Southern California Coastal Ocean Observing System (SCCOOS), to extend sampling further inshore. During the winter and spring cruises, the survey grid typically expands further north to the San Francisco Bay region (**Figure 1**) to capture most of the sardine spawning spatial domain. The extended CalCOFI grid includes 113 stations, 16 of which fall in California state waters, 89 in U.S. federal waters, and 8 are located in international waters outside of the U.S. exclusive economic zone (**Figure 1**). Due to fluctuating funding between 1969 and 1983, for this period quarterly cruises were only conducted on a triennial basis. Line 90 represents the best sampled CalCOFI line, and spring and summer samples are more complete within the CalCOFI time-series than winter or fall (Peabody et al., 2018).

A suite of hydrographic and biological data are collected at CalCOFI stations and during transit between stations. Hydrographic bottle samples for temperature, salinity, oxygen, and phosphate were collected since 1949 at specific depths (up to 500 m). Nutrient analyses expanded to include silicate, nitrate and nitrite for bottle samples starting in 1961, and then chlorophyll in 1973, dissolved inorganic carbon (DIC) in 1983, and primary production using ^{14}C incubations in 1984. CTD profiling began in 1993, and at each station an SBE 911plus CTD is deployed to measure water column temperature, salinity, dissolved oxygen, chlorophyll *a*, and photosynthetically active radiation up to a depth of ~515 m. The CTD is an oceanographic sensor package that is surrounded by a rosette of Niskin bottles for discreet water sampling and is deployed vertically through the water column to determine the hydrographic profiles of elements of interest (Williams, 2009). An SBE 18 pH sensor was added to the CTD sensor package starting in 2009.

Biological sampling at stations is conducted predominantly with nets. Vertical, oblique and surface tows are routinely collected. Oblique tows are a consistent feature of CalCOFI, and a tow is conducted at each station to sample the plankton. In 1978, bongo nets replaced ring nets to improve sampling, and an adjustment is applied for abundance estimates for some

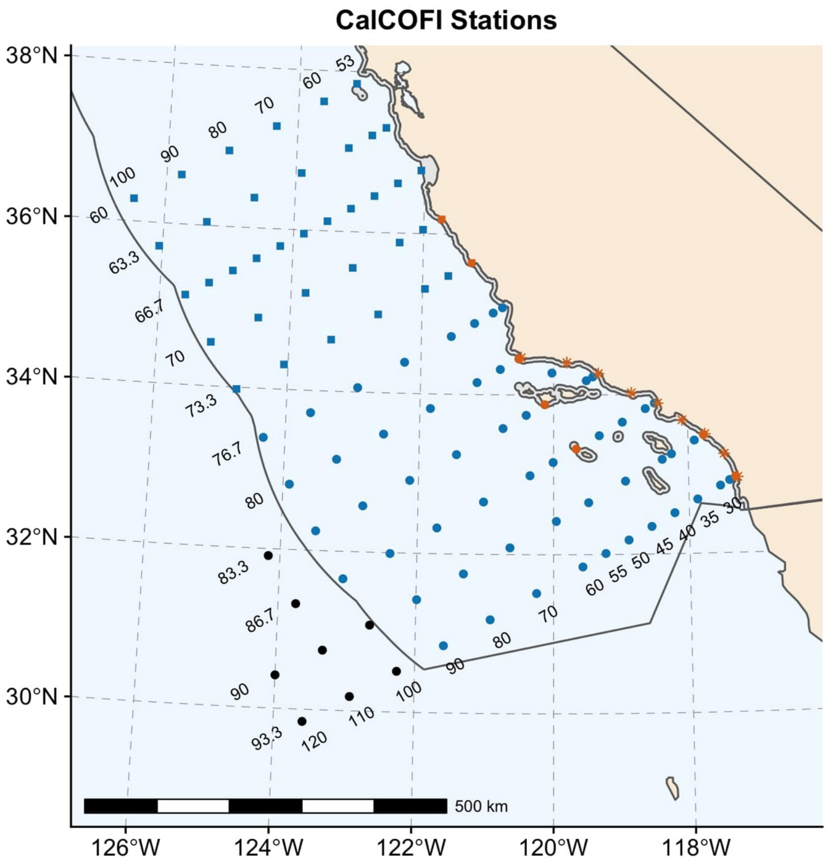

FIGURE 1 | Map of 113 CalCOFI stations as relates to U.S. jurisdictional boundaries. In CalCOFI stations are shown based on if they occur in state (orange), federal (blue), or international (black) waters. The US exclusive economic zone is shown with a black outline and California state waters are shown in gray surrounding land (tan). Circles represent the 66 station "core grid," squares represent the 38 extended grid stations, and the 9 nearshore stations are represented with asterisks. CalCOFI lines (horizontal) and stations (vertical) are indicated by the labels on the left of the grid and at the top and bottom of the grid, respectively. Not all station labels are shown. A scale bar is provided for spatial reference.

fish and zooplankton taxa to correct for changes in catchability (Ohman and Lavaniegos, 2002; Thompson et al., 2017b). A flow meter is attached to the net allowing for a measurement of sampling volume as the bongo net is towed obliquely from 212 m to the surface at 1–2 knots. Zooplankton displacement volume is measured for each sample, and all ichthyoplankton are removed. Ichthyoplankton are then identified to lowest possible taxon and enumerated. The ichthyoplankton time series is one of the best studied data products from the CalCOFI Program and includes over 400 species of fish. In recent years, certain invertebrate species of commercial importance have also been sorted and time series are available since 1997 for Market Squid (*Doryteuthis opalescens*) paralarvae and since 2007 for Spiny Lobster (*Palinuridae* spp.) phyllosoma. Ichthyoplankton and zooplankton from each tow are preserved in 5% formalin or 95% ethanol and kept in collections. Genetic barcoding has been conducted to identify species such as some rockfishes from ethanol-preserved samples that are difficult or impossible to identify based solely on morphology (Thompson et al., 2016, 2017a). Additionally, zooplankton are sampled using vertically towed PairoVET nets deployed to 70 m depth, and manta nets (1977–present), which are towed at the surface. Information about net sampling procedures are presented in Kramer et al. (1972) and Smith and Richardson (1977). All hydrographic[1] and ichthyoplankton data[2] are quality-controlled and publicly available; invertebrate zooplankton data are partially available[3].

Larval abundances from the CalCOFI ichthyoplankton time-series are used to estimate or index adult spawning stock biomass. Egg and larval abundance data have been used routinely in conjunction with trawl surveys to estimate the spawning stock of Northern Anchovy (*Engraulis mordax*) and Pacific Sardine (*Sardinops sagax*) via the daily egg-production method (Lasker, 1985). In the absence of additional trawl data, larval abundance data alone have been used to provide a coarser index of adult spawning stock biomass for Northern Anchovy and Pacific Sardine (Koslow et al., 2011), rockfish (*Sebastes* spp.; Moser et al., 2000), California Halibut (*Paralichthys californicus;* Moser and Watson, 1990), and for other rocky nearshore taxa (Moser et al.,

[1] https://new.data.calcofi.org

[2] https://upwell.pfeg.noaa.gov/erddap

[3] https://oceaninformatics.ucsd.edu/zoodb

2000). The fact that larval fish are predominantly sampled during the preflexion stage when net-sampler avoidance is minimal and post-spawning losses to mortality are still low relative to later larval stages makes them an effective index of spawning stock biomass (Koslow et al., 2018). However, ichthyoplankton counts likely do not correlate with recruitment success due to high and variable mortality at the larval stage. CalCOFI samples are well suited to document dynamics of hundreds of species of fishes because although adult populations reside in different habitats (e.g., pelagic, mesopelagic, benthic), larvae of almost every species are found in the upper 200 m of the water column (Moser, 1996; Moser and Watson, 2006). Thus, the program focuses its efforts on pelagic sampling and no benthic samples are collected.

Since 2014, CalCOFI has incorporated genomic sampling through collaboration with scientists at the J. Craig Venter Institute, La Jolla, CA, United States. Seawater samples at specific stations along the CalCOFI grid are collected and sequenced for RNA and DNA to study the microbial and eukaryotic community composition and gene expression in the water column. This is called the NOAA CalCOFI Genomics Project.

A series of underway samples are also collected on CalCOFI cruises which include ADCP measurements, meteorological data, and marine mammal and seabird observations. Since 1996, the Continuous Underway Fish Egg Sampler (CUFES) (Checkley et al., 1997, 2000) has been used to collect continuous underway data from which Northern Anchovy, Pacific Sardine, Jack Mackerel (*Trachurus symmetricus*), Pacific Hake (*Merluccius productus*), and squid (*Decapodiformes* spp.) eggs are identified. Additional autonomous tools have also expanded the scope of CalCOFI sampling. Since 2005, underwater spray gliders collect hydrographic data, including temperature and salinity, along CalCOFI lines, which have included lines 66.7, 80, 90, and 93 (Davis et al., 2008). Dissolved oxygen sensors are also now being deployed on spray gliders. Additional details on the CalCOFI sampling methodology that are outside of the scope of this study can be found in McClatchie (2014).

Looking Back at the Past 70 Years and Forward to the Future

To examine the CalCOFI record in relation to future changes predicted by climate models, we first compare the length and trends of the CalCOFI time series for climate relevant variables to that of the Keeling Curve (**Figure 2**). The Keeling Curve is a record of atmospheric carbon dioxide measurements taken at the Mauna Loa Observatory in Hawaii since 1958, and has been fundamental to tracking anthropogenic greenhouse gas forcing and the subsequent relationship to climate change (Keeling and Keeling, 2017). For the CalCOFI time series, we examine trends in winter sea surface temperature (SST) (**Figure 2B**) and winter dissolved oxygen concentration at 200 m (**Figure 2D**) across the core 66-station CalCOFI grid. For ocean acidification-relevant measurements, we synthesize a timeline of existing measurements (**Figure 2C**). As an example of the ichthyoplankton dataset, trends in the annual mean densities of Panama Lightfish, *Vinciguerria lucetia*, which is a warm-affinity mesopelagic species, are calculated across the core 66-station CalCOFI grid (**Figure 2E**).

We then compare climate trends observed from the CalCOFI dataset with expected warming predicted from a climate model. We use the Large Ensemble Community Project (LENS) (Kay et al., 2015) to look at predicted changes in annual mean SST, pH, and oxygen concentration for the CCE under the "business as usual" emissions scenario (RCP 8.5). LENS includes a 40-member ensemble of fully-coupled Community Earth System Model version 1 (CESM1) simulations for the period 1920–2100. From 1920 to 2005, the simulations use historic emissions data and after 2005 use RCP 8.5 projected emissions (Kay et al., 2015). We selected 1950–2100 as the period of interest to capture the period which corresponds to the CalCOFI time series.

Identifying Management and Stakeholder Use of CalCOFI Data

To evaluate how CalCOFI data currently informs fisheries and ecosystem management, we examine the fisheries and ecosystem management structures for the US West Coast, including federal and state policies, and identify instances where CalCOFI data are used for informing management. We also examine the spatial overlap between the CalCOFI grid and relevant management jurisdictions including state, federal, and international waters, and different marine conservation areas (California Marine Protected Areas, National Marine Sanctuaries, and rockfish conservation areas). We then identify stakeholder user groups of CalCOFI data, by considering both known users (i.e., those who have published CalCOFI data or have actively used it for their organization) and potential users (i.e., new organizations or groups that have not used CalCOFI data but whose activities overlap with the CalCOFI grid). Based on their known activities, we then consider the specific needs of different stakeholder groups related to questions of climate change impacts and adaptation, and where possible identify opportunities for better aligning stakeholder needs with CalCOFI monitoring.

By evaluating the climate change-relevant data collected by an ocean observing program and considering it within the landscape of ocean management policies and spatial overlap with regional stakeholders, other oceanographic observing programs can follow this process to better link their basic monitoring data to stakeholder needs relating to climate change adaptation.

RESULTS

An Overview From CalCOFI Data Regarding Marine Ecosystem Responses to Climate Forcing

Ocean warming, acidification and deoxygenation are some of the main climate change stressors for marine ecosystems (Bopp et al., 2013; Pörtner et al., 2014; Henson et al., 2017). In addition to anthropogenic climate change, strong multidecadal natural climate variability affects the California Current region, which can be seen in the CalCOFI time series (**Figure 2**).

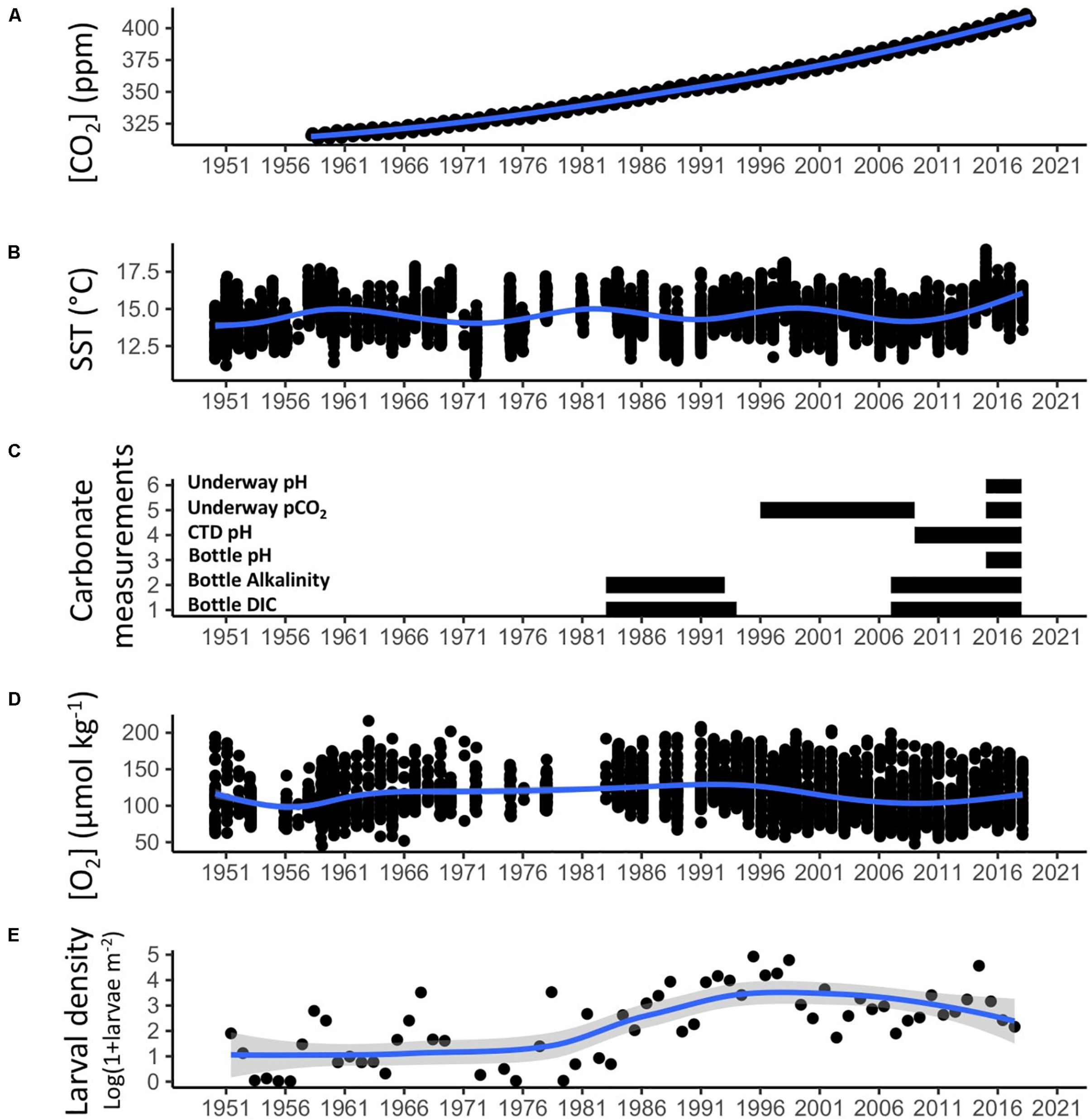

FIGURE 2 | The CalCOFI time-series is a Keeling Curve for the California Current Ecosystem. The Keeling Curve **(A)** shows the increase in atmospheric CO_2 at Mauna Loa through the twentieth and twenty-first century and is an essential record for understanding climate forcing. The CalCOFI record **(B–E)** extends a similar length and is an essential record of how the ocean responds to climate forcing through hydrographic and biological changes. Unlike the atmospheric record, the oceanic record is noisy and shows pronounced multi-decadal natural variability. Trends in winter sea surface temperature (SST) across the core 66-station CalCOFI grid are shown in **(B)**, the extent of all carbonate chemistry measurements taken on CalCOFI cruises is shown in **(C)**, trends in winter dissolved oxygen at 200 m across the core 66-station CalCOFI grid are shown in **(D)**, and trends in the annual mean density of Panama Lightfish, a warm-affinity mesopelagic species, across the core 66-station CalCOFI grid is shown in **(E)**. Blue trend lines show smoothed conditional means.

One of the dominant modes of low-frequency variability (∼30 years) that affects the CCE is the Pacific Decadal Oscillation (PDO) which is based on variability of sea surface temperatures in the North Pacific (Mantua et al., 1997). Negative phases of the PDO are associated with colder, more productive overall conditions in the CCE, and positive phases are associated with warmer, less productive conditions. The North Pacific Gyre Oscillation (NPGO) is also a dominant mode of low-frequency variability and is significantly correlated with multiple oceanographic conditions in the CalCOFI region, including upwelling strength, nutrient fluxes, and sea surface salinity anomalies (Di Lorenzo et al., 2008). Higher frequency variability (2–7 years) is driven by the El Niño Southern Oscillation, which was first recognized as a Pacific-wide phenomenon at a CalCOFI

Symposium (Sette and Isaacs, 1960). El Niño conditions are characterized by warm, less productive conditions in the CCE, while La Niña conditions are characterized by colder, more productive conditions. Stronger and more frequent El Niño conditions occur during positive PDO phases, while La Niña conditions are more pronounced and frequent during negative PDO phases. The CalCOFI record captures a negative PDO phase which lasted from 1950 to 1976 and a positive PDO phase that commenced after 1976. Although it was speculated that the PDO returned to a cool phase following a strong La Niña in 1999 (Zwolinski and Demer, 2012), the string of mostly negative PDO values between 1998 and 2003 now appear to have been more of a transient event rather than an actual regime shift (Bond et al., 2003). Consistent with this, there was no evidence of a prolonged change in the CalCOFI ichthyoplankton assemblage after 1998 (Peabody et al., 2018) and record high water temperatures occurred between 2014 and 2016 (Jacox et al., 2018). Therefore, many studies consider the CalCOFI record in terms of the "cold" (1950–1976), "warm" (1977–1998), and present (1999-present) period.

Anthropogenic climate change is a directional change on which these natural modes of variability are layered, and the high amount of natural variability in the system makes it difficult to discern the anthropogenic signal (**Figure 2**). However, projections of the climate in the North Pacific for the twenty-first century predict an increasing importance of anthropogenic warming over internal modes of variability (Bond et al., 2003; Di Lorenzo et al., 2008; Bonfils and Santer, 2011). Here, we briefly review how the CalCOFI record has informed our understanding of how the SCCE responds to different timescales of climate variability.

Warming

Despite strong inter-annual variability, temperatures within the CalCOFI region have shown a secular warming trend from 1949 to 2000, with stronger warming observed inshore than offshore (McClatchie, 2014). From 1949 to 2000, sea surface temperatures (SST) warmed by 1.3°C or 0.025°C per year on average (Di Lorenzo et al., 2005). The upper 100 m of the water column warmed 0.8°C during the period 1950–1992, representing an average warming of 0.019°C per year (Roemmich, 1992). Comparing the CalCOFI record with the temperature time series maintained at the Scripps pier, showed a slightly stronger warming trend inshore at the Scripps pier (Roemmich, 1992). From 1950 to 1993, the heat content integrated over the upper 200 m of the water column increased by 6.2–9.1% inshore and 2.4–7.1% offshore in the central and southern CCE (Palacios et al., 2004). From 1951 to 2001, heat content in the upper water column (20–200 m) increased with time roughly ten times more than the heat content of the deeper waters (300–500 m) (Hsieh et al., 2009).

Ocean Acidification

In the CCE, changes in carbonate chemistry can be driven both by changing water mass properties and circulation patterns (Meinvielle and Johnson, 2013), as well as the addition of anthropogenic CO_2 from the atmosphere into the water (Bednarsek et al., 2014). McClatchie et al. (2016) examined changes in carbonate system parameters using the empirical equations derived for the California Current region (Alin et al., 2012). Near the coast, more acidic conditions were found at shallower depths than at CalCOFI stations offshore; the aragonite saturation horizon ($\Omega = 1$) depth ranged from 84 to 267 m offshore and 50 to 250 m inshore (McClatchie et al., 2016). From 1985 to 2011, an acidification trend was detected; the average aragonite saturation horizon ($\Omega = 1$) depth shoaled by 18 m, and an 11% increase in acidity was observed at nearshore stations for depths between 15 and 100 m (McClatchie et al., 2016). This analysis likely underestimated acidification because it did not incorporate changes due to the rising atmospheric CO_2 concentration. For the California Current region, Bednarsek et al. (2014) estimated a potential anthropogenic contribution of 1.19 μmol kg^{-1} yr^{-1} to source-water DIC that is upwelled off California, based on an increasing trend in North Pacific surface water DIC.

While a continuous time-series of inorganic carbon system measurements is not available from CalCOFI to study ocean acidification (OA) trends, a multi-decadal record of measurements exists (**Figure 2C**). From 1983 to 1994, DIC measurements from ~10 m depth were collected by Charles David Keeling from a number of CalCOFI stations. Since 2008, DIC measurements have been collected at certain CalCOFI stations at depths between 0 and 200 m and analyzed by Andrew G. Dickson's lab at Scripps Institution of Oceanography, La Jolla, CA. Due to historical factors, Line 90 in the CalCOFI grid has the best coverage for DIC measurements. Alkalinity measurements of CalCOFI bottle samples have also been made during the periods 1984–1993 and 2007-present, and more recently pH measurements have been made from 2015-present. Additionally, underway pCO_2 measurements were conducted in collaboration with MBARI scientists from the mid 1990s to late 2000s on CalCOFI cruises, and more recently are being collected by Todd Martz's group at Scripps Institution of Oceanography, La Jolla, CA, since 2015. Since 2009, a pH sensor has also been added to the CTD package. Integrating these various carbonate system measurements across the CalCOFI time series to look at spatial and temporal OA trends is an area of current research.

Ocean Deoxygenation

Oxygen levels in the ocean are declining (Schmidtko et al., 2017; Breitburg et al., 2018), driven by changes in ocean ventilation, transport, and oxygen consumption linked to climate change (Levin, 2018; Oschlies et al., 2018). CalCOFI oxygen measurements are essential to understanding and resolving the oxygen dynamics in the SCCE. From 1984 to 2006, strong linear declines in oxygen were observed; declines were strongest nearshore, the largest relative declines occurred at 200–300 m depth, while the largest absolute declines occurred in the upper water column with oxygen losses of 1–2 μmol O_2 kg^{-1} year^{-1} (Bograd et al., 2008). At a nearshore CalCOFI station off San Diego, the depth of the hypoxic boundary ($O_2 < 60$ μmol kg^{-1}) shoaled by up to 90 m during this 23 years period, and shoaled by 41 m on average across the CalCOFI grid (Bograd et al., 2008).

Compared to oxygen loss observed across the world's oceans (Schmidtko et al., 2017), the SCCE has one of the most rapid observed oxygen declines. However, the unique length of the CalCOFI time-series allowed these changes to be contextualized within longer term regional trends. McClatchie et al. (2010) analyzed the extended CalCOFI time-series (1950–2007) and found that the recent linear decrease in oxygen was not representative of the whole time-series. Instead, a clear multidecadal signal emerged, with the period between 1950 and 1957 characterized by low oxygen conditions, the period between 1980 and 1987 by higher oxygen conditions, and 2000–2007 by low oxygen conditions (**Figure 2D**, McClatchie et al., 2010).

The non-monotonic nature of changing oxygen conditions in the CCE is related to natural low-frequency variability within the system (Deutsch et al., 2005). Observed declines in oxygen are thought to be associated with increased volume of advected Pacific Equatorial Water which is transported northward by the California Undercurrent (Meinvielle and Johnson, 2013; Bograd et al., 2015; Ren et al., 2018) and has a low oxygen signature. At shallower depths (<150 m), increases in respiration may also contribute to oxygen decline (Booth et al., 2014; Bograd et al., 2015; Ren et al., 2018). At shorter timescales, ENSO events affect the oxygen dynamics in the SCCE by altering the thermocline depth, with La Niña conditions characterized by lower oxygen and pH conditions (Nam et al., 2011). In general, any declines in oxygen are also coupled with decreases in pH; thus, deoxygenation and acidification will be simultaneous stressors for marine organisms in the CCE (Levin, 2018). Due to the high level of natural variability of oxygen in the CCE, forced declines in oxygen due to secular climate change are only predicted to emerge by ∼2030 (Long et al., 2016). The CalCOFI dataset is essential to ongoing monitoring of deoxygenation trends in the SCCE.

Ecosystem Responses to Climate Forcing

Marine communities are not static, but change through time in response to climate forcing. Changes can manifest through shifting species composition, changes in abundance, diversity, distribution, and trophic interactions (Poloczanska et al., 2016; Griffiths et al., 2017). Since CalCOFI collects both hydrographic and biological samples, it offers insight into how the SCCE responds to climate forcing at different timescales.

The CalCOFI ichthyoplankton time series has revealed that the fish community exhibits changes in distribution, abundance, and diversity in relation to climate forcing (Hsieh et al., 2005, 2008, 2009; Koslow et al., 2011, 2013, 2015, 2017, 2018; McClatchie et al., 2016, 2018; Peabody et al., 2018; Siegelman-Charbit et al., 2018). Hsieh et al. (2008, 2009) found that from 1951 to 2002, oceanic and coastal fishes in the SCCE shifted their distribution poleward in response to warming, with diel-vertical migrating species showing higher responsiveness to environmental change than non-migrating species (Hsieh et al., 2009). Abundance of oceanic species was significantly positively correlated with temperature, irrespective of if fish were warm water, cold water, or broadly distributed (Hsieh et al., 2009). Hsieh et al. (2009) posit that intensified stratification observed in the SCCE during the warm period created a more suitable habitat for oceanic species, thus leading to a spatial distribution shift shoreward during this period, and higher abundances within the CalCOFI grid. Similar trends were found when analyzing relationships across interannual and multidecadal scales, suggesting that these relationships may also manifest with secular climate change.

Trends in the CalCOFI ichthyoplankton time-series have also been compared to an independent time-series of ichthyoplankton collected from California power plant cooling water intakes in Southern California. Siegelman-Charbit et al. (2018) found that the abundances of coastal cold-water affinity fish taxa in the SCCE declined dramatically from 1970 to the 2000s, and this decline was related to ocean warming, increased coastal upwelling, reduced offshore upwelling and advection, as well as a decline in productivity and zooplankton prey availability. Koslow et al. (2015) also reported declining larval fish abundances across the same period, driven by declines in Pacific hake, Northern anchovy, and Pacific sardines. These results contrast with findings by Hsieh et al. (2009) for oceanic fish species, and indicate that oceanic and coastal species respond differently to climate forcing in the SCCE.

In general, warm periods in the CalCOFI record are associated with increases in fish species richness due to an influx of tropical or warm-water affinity species from the south (McClatchie et al., 2016, 2018; Koslow et al., 2017). This trend has been seen for both the 1997/1998 El Niño as well as during periods associated with increased intrusion of warm southern water (1985–1996 and 1999–2014) (McClatchie et al., 2016). Species richness increased despite the more acidic signature of southern water, suggesting that fish community species richness is not negatively impacted by current levels of acidification (McClatchie et al., 2016). Across the CalCOFI time-series (1969–2011) fish species richness is positively correlated with periods of warmer temperatures, higher midwater oxygen concentration, and the warm phase of the PDO (Koslow et al., 2017). It is worth noting that conditions that give rise to higher ichthyoplankton diversity (i.e., species richness and community evenness) are associated with poorer primary productivity conditions, and thus correspond to lower fisheries productivity (Koslow et al., 2017).

Analyses of the CalCOFI ichthyoplankton time-series have also informed our understanding of warm and cold-associated fish species (Hsieh et al., 2005; McClatchie et al., 2016, 2018). Mesopelagic cool-assemblage species include the Northern Lampfish (*Stenobrachius leucopsarus*) and California Smoothtongue (*Leuroglossus stilbius*), while mesopelagic warm-assemblage species include the Mexican Lampfish (*Triphoturus mexicanus*) and the Panama Lightfish (*Vinciguerria lucetia*) (McClatchie et al., 2016). Warm assemblage species tend to increase in abundance during warm events (McClatchie et al., 2016; Koslow et al., 2018) and these species have shown a secular trend of increasing relative abundance from 1951 to 2016 (McClatchie et al., 2018). From 2011 to 2015, mesopelagic fish with tropical or subtropical distributions became increasingly dominant in the SCCE (Koslow et al., 2018). Commercially important species also show relationships with climate forcing. Bocaccio Rockfish (*Sebastes paucispinis*) ichthyoplankton abundance is negatively correlated with

CalCOFI SST, while Pacific Sardine and Pacific Mackerel (*Scomber japonicus*) ichthyoplankton are positively correlated with CalCOFI SST (Hsieh et al., 2005). To examine changes in fishing pressure, an annual review of California fisheries (CDFW, 2015) can be examined.

Oxygen dynamics in the SCCE also correlate with trends in the CalCOFI ichthyoplankton time-series. Mesopelagic fish abundance showed a 63% decline between periods of high and low oxygen in the CalCOFI record, suggesting that deoxygenation may result in declining mesopelagic fish abundances in the future (Koslow et al., 2011). Lindegren et al. (2017) also found that mesopelagic fish abundance was positively correlated with oxygen concentration, and exhibited a threshold relationship. In contrast, oxygen was not found to be a significant variable in explaining trends in the abundance of zooplankton, euphausiids, pelagic fish, or Pacific hake in the SCCE (Lindegren et al., 2017). The vertical distributions of adult fish may also be affected by oxygen changes but this cannot be seen with the CalCOFI dataset, which only samples the early life history stages.

Key findings have also been made with the CalCOFI zooplankton dataset. Between 1951 and 1993, zooplankton displacement volume decreased significantly, correlated with a warming of surface waters (Roemmich and McGowan, 1995). From 1951 to 2005, the mean zooplankton displacement volume decreased by 64% off southern California and 68% off central California, which was attributed to a decline in high volume-low carbon zooplankton (i.e., "cool phase" salps) in the latter half of the time-series (Lavaniegos and Ohman, 2007). Zooplankton biomass in terms of carbon showed no long-term trend, highlighting that the zooplankton displacement volume is not directly correlated with zooplankton biomass in terms of carbon (Lavaniegos and Ohman, 2007). Planktonic copepods, followed by euphausiids, dominate the carbon biomass in the zooplankton samples (Lavaniegos and Ohman, 2007). The zooplankton data have also shown that certain krill species (euphausiids) have warm (e.g., *Nyctiphanes simplex* and *Euphausia eximia*) and cold (e.g., *Euphausia pacifica* and *Thysanoessa spinifera*) affinities, and that krill distributions respond to natural climate forcing through ENSO and PDO cycles (Brinton and Townsend, 2003).

Since CalCOFI samples capture species that occupy multiple trophic levels as adults, the data have been used to study how climate forcing affects food web interactions and ecosystem resilience (Lindegren et al., 2016, 2017). Despite the presence of pronounced multidecadal variability and associated changes in the abundance of certain species and groups in the SCCE, ecosystem functioning has largely remained unchanged, due to complementarity of species within trophic levels (Lindegren et al., 2016). In other words, functionally similar species have opposite responses to climate drivers, resulting in overall ecosystem stability. Bottom-up control is the primary mode of ecosystem regulation in the SCCE (Lindegren et al., 2017), meaning that changes to primary production are carried up the food chain and directly affect top predator abundances. However, under El Niño-like conditions (i.e., weak upwelling, low nutrient concentrations, and low primary production), Lindegren et al. (2017) found that bottom-up forcing is reduced in the SCCE and the system exhibits "wasp-waist" dynamics with mid-trophic levels exerting moderate top-down forcing (Cury et al., 2000). Nitrate concentration, wind stress curl-driven upwelling, and temperature were found to be the best single predictors of ecosystem state across multiple trophic levels (Lindegren et al., 2016), suggesting that climate impacts on these variables will have the greatest impact on the overall ecosystem state of the SCCE.

Regime shifts, meaning rapid, abrupt and persistent changes in ecosystem structure and function (Mollmann et al., 2015), have also been detected in the SCCE using the CalCOFI dataset. Based on the ichthyoplankton time-series, regime changes were detected in 1965 and 1976 for both the summer and spring larval fish assemblage, and additionally in 1983 and 1990 for the spring assemblage (Peabody et al., 2018). The dominant pattern since the strong 1976 regime shift has been a long-term trend toward a more southern and offshore assemblage composition in the CalCOFI region (McClatchie et al., 2018; Peabody et al., 2018). When looking at trends across multiple trophic levels in the CalCOFI dataset, significant break points were identified in 1965, 1976, 1987, and 1999 (Lindegren et al., 2016). Major ecosystem regime shifts in the CCE often occur when the PDO and NPGO show strong, simultaneous, and opposite sign reversals (Di Lorenzo et al., 2008). Therefore, climate change impacts in the SCCE will depend on the phase relationship between fluctuations of the PDO and NPGO modes (Di Lorenzo et al., 2008) along with the forcing of secular climate warming.

Future Climate Change Projections for the California Current

While CalCOFI has provided many insights into how the SCCE has responded to natural and anthropogenic climate forcing over the last 70 years, the twenty-first century will likely be defined by more rapid change and a larger contribution of anthropogenic forcing. Several valuable modeling products are available that allow for projecting future conditions in the California Current under different climate scenarios. However, developing accurate models at timescales that matter to resource managers is still an active area of research. Some of the key challenges involve: properly reproducing biogeochemical dynamics, which are essential for projecting changes in nutrients, oxygen, and pH; improving model resolution, which is important for resolving processes such as upwelling at smaller spatial scales; and accurately reproducing coastal dynamics, which are key for nearshore species such as Dungeness Crab (*Metacarcinus magister*) and California Spiny Lobster (*Panulirus interruptus*) (Holt et al., 2009, 2014; Stock et al., 2011). For managing some living marine resources, climate projections on the 3–10 years time scale are the most useful, however, these timescales are not well represented within global climate and ocean models (Busch et al., 2016).

From 1950 to 2019, the Large Ensemble Community Project (LENS) model (Kay et al., 2015) predicted an average increase in SST by 0.5°C, a decrease in surface pH by 0.1, and a decrease in surface oxygen concentration of 2.56 mmol m^{-3} for the SCCE. By 2050, an additional 1.25°C warming, a 0.1 surface pH decrease, and a 5.75 mmol m^{-3} decrease in surface oxygen concentration is predicted under the RCP 8.5 "business as usual" emissions scenario (**Figure 3**). By end of century (2100), conditions in the CalCOFI region are projected to be 3.2°C warmer (SST), surface pH will be 0.3 units lower, and surface oxygen concentrations will have decreased by 13.83 mmol m^{-3} compared to 2019 conditions under the RCP 8.5 emissions scenario (**Figure 3**).

The CalCOFI time series shows a stronger warming trend than that projected by the LENS ensemble from 1950 to 2019. While the LENS ensemble showed an average increase in SST by 0.5°C, CalCOFI observations show an average SST warming of 1.45°C for the core CalCOFI grid. For 1950–1952, the average SST across all CalCOFI core stations was 15.16°C (ranging from 10.11 to 21.14°C), while in 2016–2018, average SST in the same region was 16.61°C (ranging from 11.20 to 23.23°C). Thus, observations show a warming trend between ~1 and 2°C for the mean, minimum and maximum SST values observed across the time series. Thus, the LENS ensemble predictions for warming appear to be conservative compared to CalCOFI observations.

Gruber et al. (2012) and Hauri et al. (2013) used a regional ocean modeling system (ROMS) model at eddy-resolving resolution for the CCE to look at how OA may progress from 1995 to 2050. Under the high emission scenario, which represents 541 ppm P_{CO2} by 2050, the mean annual pH in the CCE decreased by 0.2 pH units by 2050, compared to pre-industrial conditions, and the aragonite saturation horizon shoaled by ~250–300 m (Gruber et al., 2012). During the summer upwelling season, low pH conditions were prominent, and by 2050 under the high emission scenario, large stretches of the nearshore central CCE became undersaturated with respect to aragonite (Gruber et al., 2012). These findings suggest that OA will progress rapidly in the CCE, and that nearshore areas (<10 km) are more susceptible to low pH and undersaturated conditions (Gruber et al., 2012). Under current conditions in the CCE, there is a strong positive correlation between the proportion of pteropods with severe shell dissolution and the percentage of water in the upper 100 m that is undersaturated with respect to aragonite (Bednarsek et al., 2014). Therefore, habitat suitability for animals that are sensitive to undersaturated conditions is predicted to decline in the CCE under climate change scenarios (Bednarsek et al., 2014).

Based on observed trends of rapid oxygen loss in the SCCE (Bograd et al., 2008, 2015), predicting future oxygen changes is an area of management interest. A recent study found an intriguing statistical relationship between subsurface anomalies in the core of the North Pacific Current and decadal changes in oxygen in the CCE (Pozo Buil and Di Lorenzo, 2017). Based on ocean observations and an ocean reanalysis product for the period 1950–2010, they found that anomalies in the North Pacific can be used to accurately predict oxygen trends on the US West Coast over a time-scale of 9–12 years and predicted a new period of strong oxygen decline by 2020 (Pozo Buil and Di Lorenzo, 2017). If accurate, predictions at these time-scales could be helpful for guiding management decisions if there exist known relationships between the abundance of fisheries species and environmental oxygen conditions. Trends in salinity showed high correlation with trends in oxygen, suggesting that salinity can be used as a passive tracer of circulation-driven changes in the oxygen dynamics (Pozo Buil and Di Lorenzo, 2017), in contrast with changes due to warming or biological use, which may become more pronounced with climate change in the CCE by year ~2030 (Long et al., 2016).

Reduction in primary production is an additional climate change stressor predicted to impact marine ecosystems (Bopp et al., 2013). However, there are large regional differences and high levels of uncertainty relating to projected changes in primary production across Earth system models (Steinacher et al., 2010). In eastern boundary upwelling systems (EBUS) such as the CCE, changes in upwelling-favorable winds may drive geographic and temporal changes in primary productivity. A meta-analysis found that upwelling-producing winds have strengthened over the past ~60 years in three EBUS including the California Current (Sydeman et al., 2014). Coastal upwelling-favorable winds have intensified and will continue to do so in poleward portions of EBUS (Rykaczewski et al., 2015). A climate model ensemble further predicts seasonal variability in CCE upwelling trends with intensification occurring in the spring but weakening in the summer (Brady et al., 2017). Changes in nutrient supply could also modulate productivity: Rykaczewski and Dunne (2010) explored this question using a biogeochemistry-coupled earth system model, showing increases in nitrate supply and productivity for the CCE under climate change, despite increases in stratification and potential dampening of vertical transport. While increased primary production has positive, bottom-up effects for fisheries production, these changes could be accompanied by decreases in oxygen and pH (Rykaczewski and Dunne, 2010).

If upwelling patterns follow current trends and predictions, there may be spatially heterogeneous variability in patterns of ocean warming and primary production in the future, though the CCE is particularly uncertain (Wang et al., 2015). Increased warming and stratification offshore would result in decreased primary production, while the influx of cold, nutrient-rich water due to increased coastal upwelling could mitigate warming and actually increase primary production near the coast (Garcia-Reyes et al., 2015). A pattern of warm offshore waters coupled with cool, productive coastal waters was observed in the central CCE in 2015 (Santora et al., 2017). This unusual oceanographic state resulted in a unique assemblage where subtropical mesopelagic taxa as well as taxa such as rockfishes that thrive under cooler, productive conditions were both very abundant in the same region, and overall species richness was the highest recorded since sampling began in 1990 (Santora et al., 2017).

Fisheries Management on the US West Coast and CalCOFI

To effectively explore how monitoring data can best be incorporated into management decisions from the perspective of climate change adaptation, a basic understanding of the structure of the policy and management framework is necessary. For the US West Coast, several jurisdictional management bodies and policies guide marine resource management. This section briefly discusses how the data collected by CalCOFI fits within the management priorities for the US West Coast.

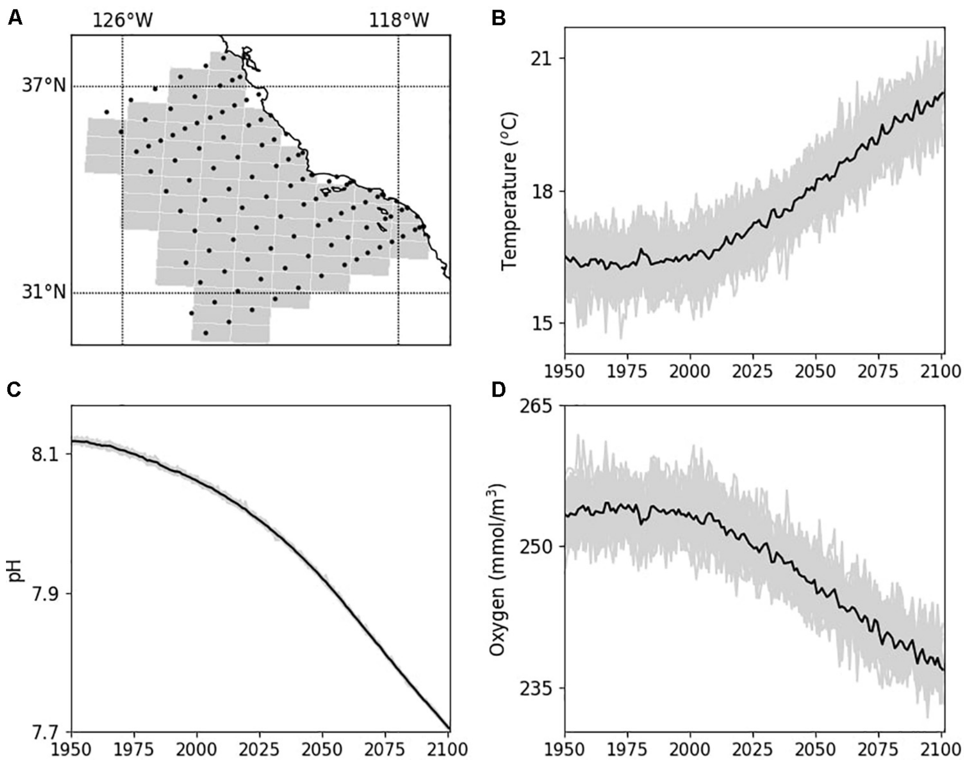

FIGURE 3 | End of twenty-first century climate projections for the CalCOFI region **(A)** for annual mean sea surface temperature **(B)**, surface pH **(C)**, and surface oxygen concentration **(D)** forced using RCP 8.5 "business as usual" emissions scenario. For the time-series, gray lines are individual model realizations (35 for temperature and 29 for pH and O_2) while the black line is the average of all model realizations. Climate model output obtained from the Large Ensemble Community Project (LENS) (Kay et al., 2015) for shaded grid cells in **(A)**.

In the U.S., Fisheries Management Plans (FMPs) are mandated for fisheries occurring in federal waters by U.S. law under the Magnuson-Stevens Fishery Conservation and Management Act (MSA) (NMFS, 2007). FMPs are the main vehicle for managing fisheries, and rely largely on single-species stock assessments for determining stock status and setting harvest limits. Management varies based on the fisheries species, whether it is fished commercially or recreationally, and whether it is managed federally and/or by the state. Federal fisheries species are managed by NOAA National Marine Fisheries Service (NMFS) through the eight regional Fishery Management Councils established under the MSA (NMFS, 2007). The Pacific Fisheries Management Council (PFMC) is responsible for management of fisheries along the US West Coast (NMFS, 2007) and includes federal, state, and tribal partners. In California, the Marine Life Management Act (MLMA) directs the California Department of Fish and Wildlife (CDFW) and the California Fish and Game Commission to manage state fisheries sustainably including developing FMPs (CDFW, 2018).

CalCOFI already contributes directly to US West Coast fisheries population assessments and management (McClatchie, 2014) by providing the PFMC with information about fish population abundances, oceanographic conditions, and ecosystem status. CalCOFI larval abundance time-series are directly used as indices of spawning stock biomass in the Bocaccio (He and Field, 2017) and Cowcod (*Sebastes levis*; Dick and MacCall, 2014) rockfish stock assessments, and were recently used to monitor population trends of Blue Rockfish (*Sebastes mystinus*; Dick et al., 2017). CalCOFI data are also used to quantify Daily Egg Production values that contribute to Pacific Sardine (Hill et al., 2017) and Northern Anchovy (Fissel et al., 2011; Dorval et al., 2018) stock assessments. Additionally, CalCOFI larval data track the status of harvested species such as Market Squid (Koslow and Allen, 2011) and Jack Mackerel (Thompson et al., 2018), and taxa such as mesopelagic fishes that are considered by the PFMC to be important forage species for higher trophic levels (McClatchie et al., 2018). This information is used by the PFMC to set harvest limits at levels that support sustainable fisheries.

In addition to evaluating the status of individual species, fisheries management in the United States also emphasizes holistic monitoring of ecosystem dynamics (Levin et al., 2008), and NOAA Fisheries is moving toward a next generation stock assessment approach which incorporates ecosystem links (Lynch et al., 2018). The inclusion of ecosystem factors such as oceanographic variables into single-species stock assessments, termed an Ecosystem Approach to Fisheries Management (EAFM), can allow for better stock-focused management decisions (Link, 2016). Currently, most stock assessments do not take environmental conditions into account due to the difficulty of elucidating predictable recruitment-environment relationships that explain enough recruitment variability to be useful for management (Myers, 1998; Jacobsen and McClatchie, 2013; Lindegren et al., 2013; Tolimieri et al., 2018). A global review of 1200 marine fish stocks revealed that only 24 stocks were managed in a way that incorporated ecosystem drivers such as climate (Skern-Mauritzen et al., 2016). The Pacific Sardine is one example in which oceanographic data directly informs an aspect of single-species stock assessment. Pacific Sardine stock productivity is higher under warmer conditions, so Pacific Sardine management uses SST averaged across the core CalCOFI grid to calculate the over fishing limit and the acceptable biological catch (Hill et al., 2017), with fishing effort increasing under higher temperatures. However, recent efforts to re-evaluate the temperature dependence of Pacific Sardine stock productivity concluded that the earlier positive correlation with SST may no longer be valid (Zwolinski and Demer, 2019).

Beyond a single-species focus, Fisheries Ecosystem Plans (FEPs) have been developed to guide fisheries management toward implementing ecological principles (Marshall et al., 2018). FEPs are encouraged but not mandated by federal fisheries policy, and five of eight regional fisheries management councils, including the PFMC (Pacific Fishery Management Council [PFMC], 2013), have developed FEPs. The PFMC also currently includes an Ecosystem Consideration appendix to their groundfish, coastal pelagic species, and salmon FMPs as an effort to inject monitoring data more directly into fisheries management decisions, which includes some CalCOFI data.

The CalCOFI dataset is well suited to not only inform the status of individual species but also to provide insight into the ecosystem state of the SCCE and how it responds to both short and long-term environmental forcing (McClatchie, 2014). Integrated ecosystem assessments (IEAs) have been developed by NOAA to support Ecosystem Based Management (EBM) (Levin et al., 2009), and the California Current was selected as the first region for the development of a full IEA, which began in 2009 and is called the California Current Integrated Ecosystem Assessment (CCIEA). CalCOFI contributes several of the indicators used by the CCIEA (**Figure 4**), including the abundances of larval Northern Anchovies, Market Squid, Pacific Hake, Jack Mackerel, Sanddabs, Pacific Sardine, Shortbelly Rockfish (*Sebastes jordani*), and a category of warmwater and coolwater mesopelagic fish larvae. Since 2013, the IEA team has presented an annual report to the PFMC on the state of the CCE including indicator trends using CalCOFI data (Harvey et al., 2018). Organizations that use the CCIEA to inform decision-making include the PFMC, the West Coast National Marine Sanctuaries, the West Coast Governors Alliance, and the North Pacific Marine Science Organization (Samhouri et al., 2013).

How Can CalCOFI Support Stakeholder Needs for Addressing Climate Change Adaptation?

While the CalCOFI dataset already supports efforts at the state and federal level to consider management approaches that are adaptive to climate change, progress can be made in linking monitoring data with actionable strategies for climate change adaptation. One approach to this is through recognizing and meeting the needs of stakeholders concerned with climate change adaptation. Here we consider several different stakeholder user groups of CalCOFI data products (**Figure 5**).

Academic Community

One of the most obvious stakeholders is the science community, which includes stock assessors, the climate modeling community, as well as scientists studying the ecological impacts of climate change. For these communities, time series and sampling consistency, attention to accuracy of measurements, rapid data availability following cruises, and easy data access are key. Given that the CalCOFI time series currently extends almost 70 years, it allows analysis of climate related interactions at multidecadal, interannual, and seasonal time scales in addition to secular (long-term) climate change. Consequently, it supports two key research needs identified by the 2009 National Climate Assessment report: improving understanding of the climate system and drivers, and improving understanding of climate impacts and vulnerability. Survey data are essential for validating regional ocean models, Global Circulation Models, and Global Climate Models and CalCOFI data are integrated into regional modeling efforts in terms of model calibration, data assimilation, hindcasting and reanalysis model products. Data-assimilative ROMS which incorporate CalCOFI data have been developed and are used by research groups at the University of California, Santa Cruz and the University of California, Los Angeles.

Fisheries and Aquaculture Community

The management community is also a key stakeholder of CalCOFI data products. Improving the paths by which monitoring data are used by management bodies will improve the ability for managers to make decisions that are more responsive to climate change. Answering how climate-related effects can be incorporated into adaptive marine resource management processes is a key question for NMFS (Busch et al., 2016), and the PFMC is currently considering this as part of its Climate and Fishing Communities Initiative.

Similarly, state management agencies such as CDFW are stakeholders of CalCOFI data products. CalCOFI provides fishery-independent time series that can be incorporated into management for key state-managed fisheries, such as California Spiny Lobster and Market Squid (Koslow and Allen, 2011; Koslow et al., 2012), and the nearshore sampling stations provide a look at more nearshore changes that are relevant to the state. There are no fishery-independent time series for many recreational fisheries, and CalCOFI data are well suited to inform stock status for taxa such as Saltwater Basses (*Paralabrax* spp.) that may be overfished (Jarvis et al., 2014). Early life history stages of these forage species are captured in the CalCOFI dataset, providing an existing time series index of spawning stock biomass for reference.

Climate change can also lead to fish stocks shifting their distribution, which can have important management implications regarding allocation. When a stock is shared across national boundaries, this can add additional complexity. The

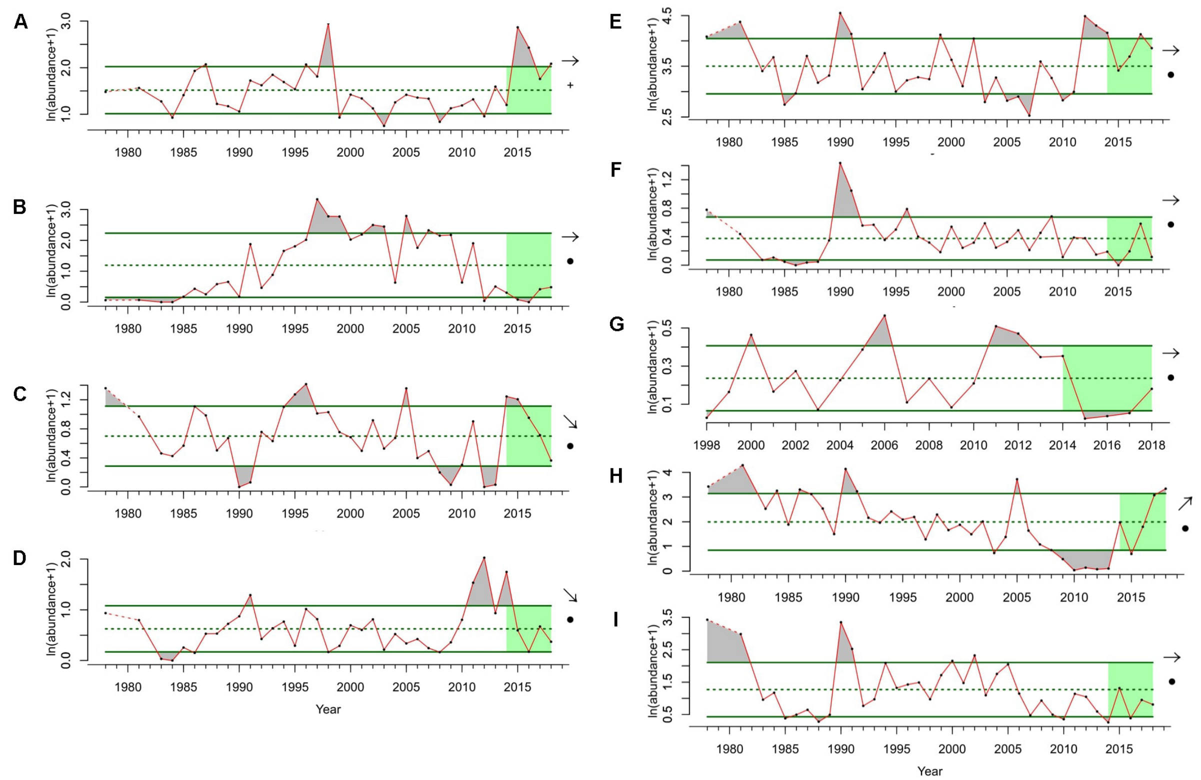

FIGURE 4 | Indicator taxa from the CalCOFI time-series that are used for the California Current Integrated Ecosystem Assessment (CCIEA). On left **(A–D)** are groups or species that exhibit a warm-affinity and may increase in total and relative abundance in the future: **(A)** warm-water mesopelagic fish larvae, **(B)** Pacific Sardine, **(C)** Jack Mackerel, and **(D)** Sanddabs. On right **(E–I)** are groups or species that exhibit a cold-affinity and may decrease in total and relative abundance in the future: **(E)** cold-water mesopelagic fish larvae, **(F)** Shortbelly Rockfish, **(G)** Market Squid, **(H)** Northern Anchovy, and **(I)** Pacific Hake. Time series come from spring CalCOFI surveys (1978–2018) on lines 76.7–93.3 and stations 28.0–120.2. Larval fish data are summed across all stations and units are in number of larval fish under 10 m^2 of surface area, transformed using ln(abundance + 1). Dotted line shows long term mean, solid green lines show standard deviation, and recent 5 years trend for each taxa is indicated by the arrow on the right of each time-series. Green shading indicates the last 5 years of the time-series.

southern end of the CalCOFI domain samples along the US-Mexican border meaning that it can provide information on shifting distributions across international boundaries. Data from the Mexican exclusive economic zone are more limited, although data from IMECOCAL (Baumgartner et al., 2008) and some catch data from Mexico are incorporated into the Pacific Sardine stock assessment. In contrast to shared US-Canadian stocks, no explicit fisheries stock sharing agreements exist between the US and Mexico. One potential emerging fishery of interest is the Humboldt Squid, which is fished in Mexican waters but is only periodically caught off California. Humboldt Squid appears to expand its range northward during warm, low oxygen periods (Gilly et al., 2013) and could emerge as a valuable fishery in a warmer future. Since commercial fishermen need specific permits to allow for fishing new species, early indicators of a shift in distribution are key to supporting responsive management actions.

The fisheries industry is also a stakeholder of CalCOFI data. The CalCOFI region captures an area of great interest to the recreational fisheries industry, which is an engine of the economy in Southern California. Many recreational fisheries species are not federally managed and therefore population trends of these species are not as well known. Since CalCOFI samples the early life history stages of over 400 fish species, it provides a valuable record of fisheries species trends that receive less management attention, but may be impacted by climate change. Sport fishing organizations and recreational fishing party boats can all be thought of as potential stakeholders of CalCOFI data. CalCOFI is also important for the commercial fishing industry. The California Wetfish Producers Association pays close attention to CalCOFI data since most wetfish (i.e., sardines, anchovies, squid, and mackerels) in the state come from southern and central California. This group is actively involved with the PFMC and coastal pelagic species management, and industry-funded researchers for the California Wetfish Producers Association use CalCOFI paralarvae counts of Market Squid in their monitoring studies.

Beyond impacts on fisheries species, there is also a need to consider adaptive capacity of the fisheries industry, which depends on the types of fisheries that are present. For example, while stock assessments focus on single-species, fishermen can fish multiple species and switch effort based on both biological and economic factors. Fuller (2018) shows that fisheries in the CCE are strongly connected by human participation,

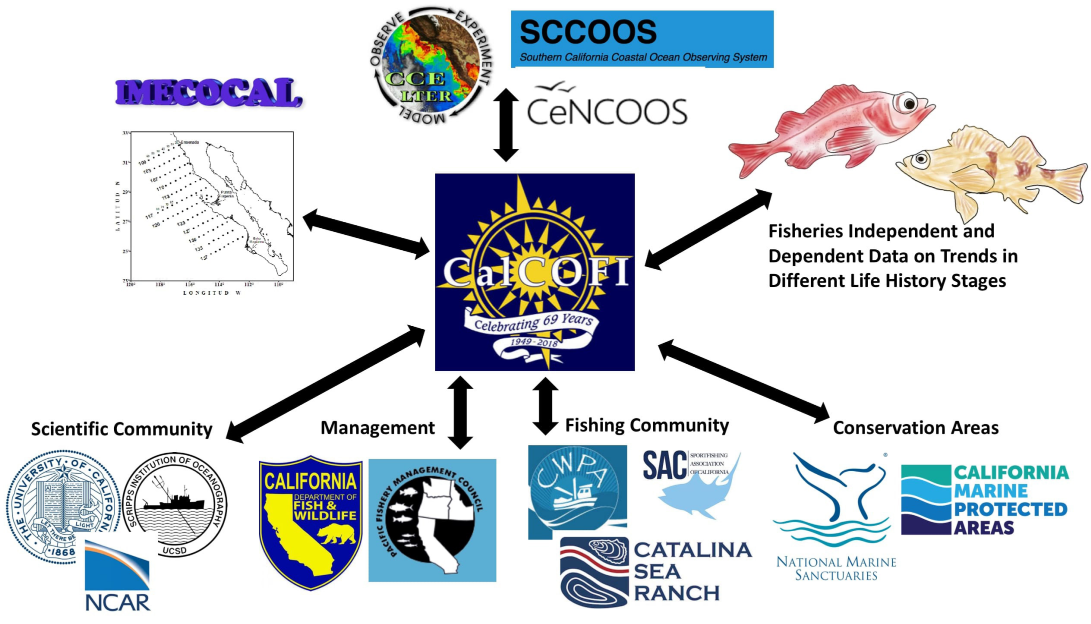

FIGURE 5 | Linking basic monitoring to decision-making for climate change adaptation requires integrating different data streams with the needs of stakeholders. CalCOFI exists alongside a wealth of additional sampling programs that, in concert, allow for comparing changes seen in the CalCOFI record across regions (upper, left), monitoring frameworks within a region (upper, middle), and sampling programs on different fish life stages (upper, right). Trends observed from these monitoring programs can support different stakeholder groups (bottom) including the scientific, management, fishing, and conservation communities on climate change adaptation strategies and actions. Arrows show that information can flow both ways.

with participants in the Dungeness Crab pot fishery showing high connectedness to many different fisheries. Incorporating considerations about effort-switching may be critical in thinking about climate change adaptation for fisheries. Since CalCOFI samples multiple species, analyses could be catered to reflect fishing practices by grouping species by connectivity of fisheries efforts, as opposed to focusing on single-species analyses or ecological groupings.

While there is currently little offshore aquaculture along the southern U.S. West Coast, there are ongoing efforts to increase its development. Brander (2010) points out that aquaculture resembles terrestrial animal husbandry more than it does wild capture fisheries, and therefore has greater scope for decision-making that is adaptive to climate change. In the future, the aquaculture industry could become an additional stakeholder of CalCOFI data. The Catalina Sea Ranch is a mussel aquaculture farm, which operates in federal waters offshore of Los Angeles, CA. It is the first aquaculture facility permitted for commercial use in federal waters in the US, and the company plans to expand its 100 acre underwater shellfish farm to 1,000 acres. Shellfish aquaculture may be especially concerned about OA impacts. There have also been proposals for finfish aquaculture in federal waters off California, such as a recent proposal by Pacific Ocean Aquafarms (previously the Rose Canyon Fisheries Farm) to raise California Yellowtail (*Seriola lalandi*) and White Seabass in offshore net pens off San Diego. While it is unclear if this venture will be approved, it is worth noting that just south of the border in Mexico, tuna aquaculture is a widespread practice, with several companies, including Baja Aqua Farms, raising tuna in offshore pens. While aquaculture groups will likely invest in their own ocean monitoring platforms, CalCOFI can provide information on larger-scale trends (i.e., changes in pH or oxygen) that are occurring independent of aquaculture practices.

Ecosystem Management Community

Planners and managers of marine conservation areas can also be considered stakeholders of CalCOFI data. Rockfish conservation areas, marine protected areas (MPAs), and national marine sanctuaries all occur within the CalCOFI region (**Figure 6**). While these conservation areas have different mandates and roles, CalCOFI can provide regional context for interpreting local trends that may be related to specific management or marine use actions. Nine CalCOFI stations occur within National Marine Sanctuaries (NMS) (seven within the Monterey Bay NMS, one within the Greater Farallones NMS, and one within the Channel Islands NMS) (**Figure 6A**). The Channel Islands NMS makes use of CalCOFI data in its formal status reports but there are clear opportunities for CalCOFI data to further inform the Monterey Bay and Greater Farallones NMS. CalCOFI stations also occur within three State Marine Reserves (South Point, Begg Rock, and Point Conception) and two State Marine Conservation Areas (Point Dume and Dana Point) (**Figure 6B**). Six CalCOFI stations also fall within the Cowcod Conservation Area (CCA) (**Figure 6C**). Data from CalCOFI stations within and outside of the CCA have been used to show trends in recovery of overfished rockfish stocks (Thompson et al., 2017a). Understanding vulnerability of marine conservation areas to climate change and considering adaptation measures

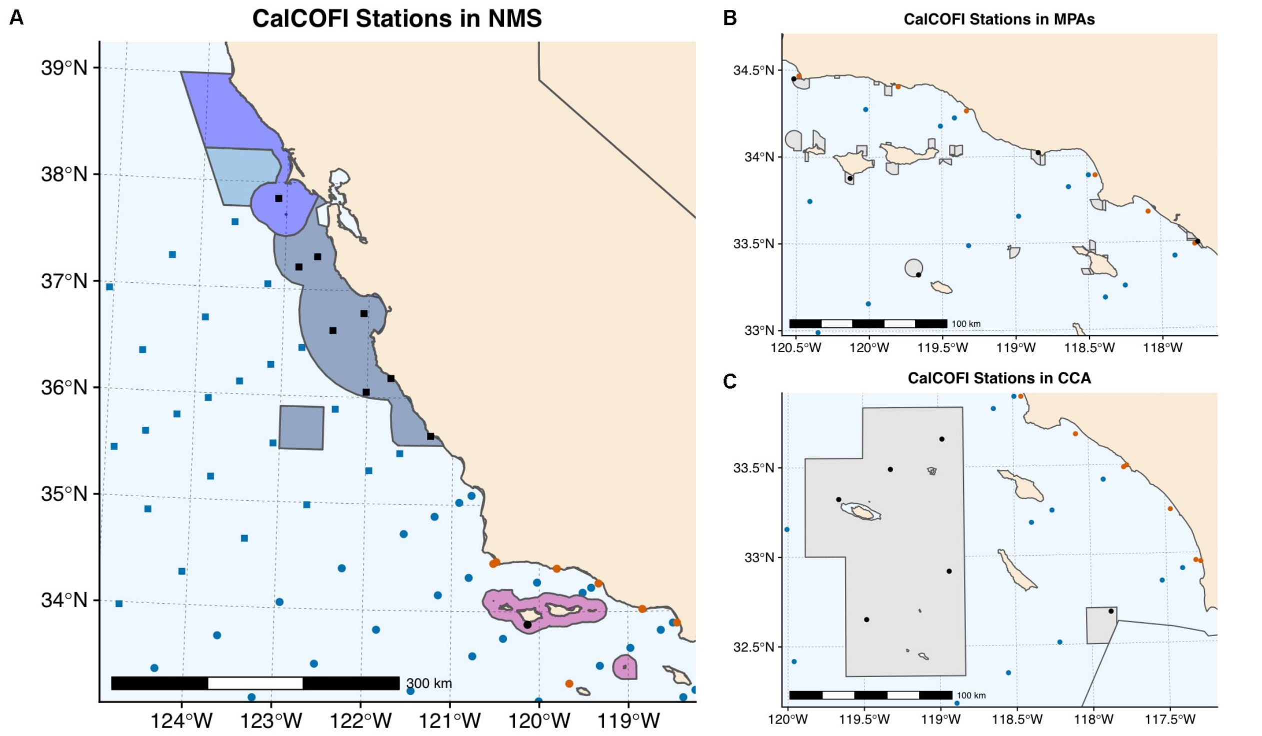

FIGURE 6 | Nine CalCOFI stations fall within the National Marine Sanctuaries **(A)**, five within Marine Protected Areas **(B)**, and six within the Cowcod Conservation Area **(C)**. CalCOFI stations that fall within conservation areas are shown in black within the conservation area boundaries. A scale bar is provided for spatial reference. Color and point type of other CalCOFI stations are consistent with **Figure 1**.

are active areas of interest for planners and managers of these areas; the CalCOFI dataset can provide important regional insight for climate change studies and opportunities to evaluate management action.

For the northern and central CCS, Sydeman et al. (2014) developed a Multivariate Ocean Climate Indicator (MOCI) to aid in the evaluation of ecosystem management decisions. The MOCI synthesizes 14 basin- and regional-scale atmospheric and oceanographic indicators from a variety of time series (Sydeman et al., 2014). If this multivariate index proves useful in guiding management decisions, a similar index could be developed for Southern California by incorporating CalCOFI data. Currently, CalCOFI scientists produce an annual State of the California Current report, which provides a review of environmental conditions in the CCE (e.g., Thompson et al., 2018). Species that are responsive to climate forcing and are dominant in abundance in the community can be good bio-indicator species and several taxa from the CalCOFI time series are used as bio-indicators for the CCIEA.

Certain non-governmental organizations such as EcoAdapt and The Nature Conservancy Climate Fisheries group are also actively involved in developing climate change vulnerability assessments and strategies for climate change adaptation for fisheries and marine ecosystem management. These groups may use CalCOFI data or products such as the State of the California Current report. Additionally, the CalCOFI Program now exists alongside a wealth of additional sampling programs that have developed in the region through time. These include: NOAA's Northwest Fisheries Science Center Groundfish Trawl Survey and the Southwest Fisheries Science Center Pelagic Juvenile Rockfish Recruitment and Ecosystem Assessment Survey and summer Coastal Pelagic Survey, the 40 + years ichthyoplankton time-series from the Southern California coastal power plant intakes (Miller and McGowan, 2013; Siegelman-Charbit et al., 2018), the CCE LTER Program, commercial and recreational fishing data (Perry et al., 2010), the IMECOCAL oceanographic survey south of the border (Baumgartner et al., 2008), the southern (SCCOOS) and central California (CeNCOOS) coastal ocean observing programs, as well as others. Leveraging knowledge, collaborations, and investment across these different groups will allow for better scientific guidance for climate change adaptation for fisheries and ecosystem management.

DISCUSSION

Climate Change Impacts on Fisheries

CalCOFI was originally developed to inform Pacific Sardine management, and continues to inform sustainable fisheries management on the US West Coast. As climate change progresses, CalCOFI can be used to monitor climate change impacts on fisheries species. The effects of climate change on fisheries production is a concern for fisheries management around the world (Barange et al., 2018; Cheung, 2018).

Climate change can affect fisheries species through direct effects of hydrographic changes (e.g., temperature, oxygen, pH) on physiology and the indirect effects of ecological changes mediated through the food web and multi-species interactions (Brander, 2010). Overall, trends indicate increased catch potential in high latitudes and decreased catch potential in lower latitudes (i.e., tropics) with climate change (Cheung et al., 2010). On the US West Coast, fisheries for Albacore Tuna (*Thunnus alalunga*) may increase with climate change and new fisheries, such as on the Humboldt Squid (*Dosidicus gigas*), may develop (Chavez et al., 2017).

Managing fisheries effectively under climate change will require the ability to identify and track trends, develop indices of change, and develop early indicators of potential shifts in biological communities, which are all areas that CalCOFI directly contributes to. Management bodies need synthesized data products, including relevant indices, to make decisions, as well as reliable climate forecasting abilities over 1–10 years time frames that guide decision-making. To do this, linkages between environmental indices and fisheries need to be effectively identified so that forecasting can be used to inform management decisions.

NOAA Fisheries has developed a Climate Science Strategy (CSS) for including climate-related information in managing fisheries and protected species, which includes conducting climate vulnerability analyses, establishing ecosystem indicators, and developing capacity for conducting management strategy evaluations (MSEs) of climate-related impacts on management targets, priorities, and goals (Busch et al., 2016). MSEs are a modeling tool which allow scientists and managers to compare the performance of different management strategies in meeting specific management objectives (i.e., fishery resource status) (Punt et al., 2016) and can be used to manage climate change risks (Plaganyi et al., 2013). To implement the CSS, regional NMFS offices develop Regional Action Plans, which are developed jointly by scientists, managers, and stakeholders. For the US West Coast, a Western Regional Action Plan has been developed (Noaa Nw/Sw Fisheries Science Centers, 2016) that lays out the NOAA Fisheries climate change strategy and the Western Regional Implementation Plan provides the means to execute the strategy. Incorporating environmental data into stock assessments to improve fisheries management under climate change is also a recognized priority for NMFS (Karp et al., 2018; Lynch et al., 2018).

While the work on climate-ready fisheries management is relatively new, several adaptation approaches have been proposed including: addressing cumulative impacts to reduce stressors for fisheries species, preparing for emerging fisheries, accounting for climate effects in stock assessments, adjusting spatial boundaries for management areas, coordinating across static boundaries such as national borders, developing regional climate change projections to guide management decisions, expanding opportunities for real-time responses to climate, and promoting social-ecological resilience (Pinsky and Mantua, 2014). Vulnerability assessments, modeling tools and MSEs, and the use of adaptation frameworks that incorporate socio-ecological considerations are all tools that can be used to address fisheries adaptation to climate change (Lindegren and Brander, 2018).

Based on CalCOFI data, in the SCCE, secular climate change will likely lead to increased fish community diversity (McClatchie et al., 2016, 2018; Koslow et al., 2017), as warm-affinity species expand their ranges, but may be associated with lower fisheries productivity for current community dominants like Northern Anchovy and Pacific Hake (Koslow et al., 2017). Changes in community composition are also expected. If conditions continue to warm, a change in dominance is expected from California Smoothtongue to Mexican Lampfish inshore and Northern Lampfish to Panama Lightfish offshore (McClatchie et al., 2016). A comparison of the 15 most abundant ichthyoplankton taxa by sampling region between the CalCOFI region and the region to the south off Baja California from the IMECOCAL Program, suggest that warming conditions may lead to increased relative abundances of Sanddabs (*Citharichthys* spp.), Pacific Sardine, and Pacific Mackerel, and decreased abundances of Pacific Hake, Rockfishes, and Croaker (*Scianidae* spp.) (Koslow et al., 2018). Additionally, decreases in primary productivity could give rise to greater influence of mid-trophic levels on ecosystem dynamics in the SCCE (Lindegren et al., 2017).

States are also increasingly interested in understanding how climate change may affect their fisheries. In California, a report on readying California's fisheries for climate change was recently developed for consideration by the CDFW to help inform the state's process to amend the Marine Life Management Act (MLMA) Master Plan (Chavez et al., 2017). CDFW has also been developing vulnerability assessments of its primary fisheries species, and the California Ocean Science Trust, Ocean Protection Council, and the Ocean and Coastal Climate Advisory Team are actively engaged in considering climate-ready management strategies in California. A challenge of considering climate change in fisheries management is that across "short" time-scales relevant to decision makers (<10 years), the anthropogenic component of climate change only adds a small increment of change compared to the natural variability experienced by fisheries species in the region (Brander, 2010). However, extreme climate events can lead to rapid effects on a fisheries species, so changes to the frequency or timing of extreme events is also a climate change vulnerability (Brander, 2010).

Species that have in the past increased in abundance during warm conditions may fare better under climate change scenarios, while those that favor cooler periods may be negatively impacted. For California fisheries, Chavez et al. (2017) categorized the following as "warm phase" species: California Halibut, Pacific Sardine, California Spiny Lobster, Pacific Mackerel, Kelp Bass (*Paralabrax clathratus*), Barred Sand Bass (*Paralabrax nebulifer*), Spotted Sand Bass (*Paralabrax maculatofasciatus*), California Sheephead (*Semicossyphus pulcher*), Kellet's Whelk (*Kelletia kelletii*), Pacific Bonito (*Sarda lineolata*), and White Seabass (*Atractoscion nobilis*). In contrast, the following fisheries species were categorized as "cool phase" species and more likely to be negatively impacted by climate change: Market Squid, Dungeness Crab, Northern Anchovy, most groundfishes, Chinook Salmon (*Oncorhynchus tshawytscha*), Pacific Geoduck

(*Panopea generosa*), Pacific Pink Shrimp (*Pandalus jordani*), Pacific Halibut (*Hippoglossus stenolepis*), and Red Abalone (*Haliotis rufescens*) (Chavez et al., 2017).

The CalCOFI dataset captures the early life history stages of many of these species, allowing for monitoring of larval relative abundance trends through time as climate change progresses. Analysis of the CalCOFI time series suggests that commercially exploited fish species show clearer distributions shifts in response to climate forcing than unexploited species (Hsieh et al., 2008), and thus may be more sensitive to climate change. The CalCOFI record has also shown that the abundance of several groundfish, such as English Sole (*Parophrys vetulus*), Cabezon (*Scorpaenichthys marmoratus*), Aurora Rockfish (*Sebastes aurora*), Bocaccio Rockfish, Shortbelly Rockfish, and other rockfishes is especially low after El Niño events (1958, 1983, 1997), likely due to the low reproductive output of the adult population for those years (Lindegren et al., 2017). Following El Niño events, both dominance and assemblage spatial structure of the ichthyoplankton community tended to return to the long-term climatological average, suggesting resilience to temporary perturbations (McClatchie et al., 2016, 2018). Given the considerable changes in warming, pH, and oxygen projected for the SCCE for the twenty-first century (**Figure 3**), as well as the uncertainty surrounding changes in primary production, the role of CalCOFI in monitoring climate change impacts in the SCCE will become even more critical.

Both "bottom up" and "top down" approaches exist to climate adaptation. One example of a "bottom up" approach is Australia's research program on Marine Climate Impacts and Adaptation (Hobday and Cvitanovic, 2017) that provides local fishermen and enterprises with knowledge and climate forecast products such as seasonal forecasts at small spatial scales. This directly provides fishermen with tools to guide their decisions. In contrast, the European Maritime and Fisheries Fund uses a "top down" approach that emphasizes strategic planning at large spatial and organizational scales (Lindegren and Brander, 2018). The US strategy for climate adaptation is more like the European approach, however, there may be opportunities to incorporate more "bottom up" elements that may increase participation from local stakeholders. Wilson et al. (2018) proposes that developing climate-ready strategies for fisheries will require increasing the level of participation in the management process by bringing together fishermen, non-governmental organizations, and traditional scientific and management bodies within an adaptive co-management framework. Using a "bottom up" approach to climate adaptation by hosting workshops or roundtables could increase participation, knowledge, and investment from the local community, and connect CalCOFI more directly with the fisheries industry stakeholders.

Limitations and Areas for Improvement

One limitation of using past trends to predict future changes is that ENSO, PDO, and NPGO climate forcing may not be representative of changes predicted with secular climate change. Koslow et al. (2018) show an intriguing recent breakdown in the observed relationship in oceanic forcing on the response of mesopelagic fish across the CalCOFI time series. While mesopelagic fish abundance was positively correlated with midwater oxygen concentration for much of the CalCOFI record, this relationship appears to be breaking down in the twenty-first century as more hypoxia-tolerant warm-water affinity mesopelagic fishes such as Panama Lightfish and Mexican Lampfish move into the region and increase in abundance, despite declining oxygen levels (Koslow et al., 2018). It is important to keep in mind that the ranges of most species extend beyond the CalCOFI grid, and therefore species abundance trends within the core CalCOFI grid do not necessarily equate to trends in overall population size (Weber and McClatchie, 2012).

The concept that environment ∼ abundance trends based on correlative data over the past 7 decades may not hold under climate change was also evident in 2018. Northern Anchovy have historically been categorized as a species that does well during relatively cold conditions while Pacific Sardine have been thought to thrive under warm conditions (Chavez et al., 2003). Biomass of adult and larval Northern Anchovy, however, approached record highs throughout the CCE from 2017 to 2018 (Thompson et al., 2018). These spawning adults had to have been born and successfully recruited during the warmest sustained waters on record between 2014 and 2016 (Jacox et al., 2018), thus challenging the notion that Northern Anchovy thrive under cold conditions. By contrast, Pacific Sardine remained at low abundances over the past decade even though ocean conditions superficially supported resurgence. These findings underscore the difficulty in predicting how species will respond to climate change and emphasize the need to understand the precise mechanisms affecting fisheries population dynamics (Checkley et al., 2017). Even with 7 decades of observations, the patterns observed provide correlations in oceanographic processes that support (but do not necessarily prove) mechanistic drivers, such as external climate forcing. For this reason the CCE LTER is built around the goal of demonstrating the mechanisms driving patterns observed in CalCOFI's observations via experimental oceanography (Ohman and Hobbie, 2008).

In this study, we identify two additional areas where CalCOFI can better support stakeholder needs. The first is developing a more robust framework for OA monitoring within the CalCOFI Program, which aligns with state needs vocalized by the Ocean Protection Council. While many carbonate system measurements have been collected on CalCOFI cruises over the years, there have been no comprehensive efforts to integrate these datasets and couple them with the biological samples. However, these samples can provide a rich baseline for studying OA trends which are predicted to manifest rapidly in the CCS (Gruber et al., 2012; Hauri et al., 2013). Knowledge of baseline carbonate system conditions may also be important in informing the development of offshore shellfish aquaculture in the region and considering future OA vulnerability for this industry. Therefore, supporting continuous OA monitoring along CalCOFI lines should be a high priority.

Lastly, one new area which CalCOFI can contribute to through the NOAA CalCOFI Genomic Project is in monitoring changes in the microbial community. There is a concern that marine infectious diseases will increase under climate change (Burge et al., 2014). Monitoring pathogen abundances and understanding conditions that give rise to outbreaks can help manage this vulnerability. HABs may also increase in frequency and severity under climate change (Wells et al., 2015) with economic implications for the fisheries, aquaculture, and coastal sectors. Since novel high-throughput sequencing approaches can provide a detailed analysis of the composition as well as activity (i.e., gene expression) of the microbial community, these novel methodologies could help inform these emerging climate change adaptation issues. Additionally, these data may be useful for environmental consulting companies that monitor sewage and powerplant outfalls, by providing a regional baseline for their samples. Systematically incorporating this sampling into the CalCOFI cruises would further the ecosystem perspective gained from the CalCOFI Program.

CONCLUSION

Frusher et al. (2014) outlined four preconditioning factors that are necessary for developing an effective climate adaptation process for fisheries management. These include: (1) early observations of rapid ocean change, coinciding with (2) observations of biological changes that provide focus for action, (3) within a region that has a strong history of marine focus and management, and (4) the presence of well-developed networks involving multiple marine resource users. Based on these, the US West Coast is well positioned to be able to respond to climate change effectively with an adaptive management framework for fisheries, aquaculture, and marine ecosystem use. The CalCOFI Program contributes in an essential way to all four of these factors, by collecting paired oceanographic and biological data on the marine ecosystem, fostering the long record and history of fisheries oceanography and studies on the effects of climate variability on marine resources for the region, and the presence of many partnerships (other scientific surveys, academic contributions, governmental scientists, and NGOs). For CalCOFI information to be most useful, managers need a fuller understanding of the linkages between biological and oceanographic changes in managed species and the whole ecosystem.

The time-series length as well as the spatial scale of CalCOFI sampling has shown that, overall, trends for warming, acidification, and deoxygenation appear to be more pronounced inshore than offshore, which suggests that coastal and nearshore ecosystems may be more vulnerable to rapid climate change. Thus, climate change effects may be felt more rapidly within state waters than federal waters, and nearshore marine protected areas may also experience more rapid change than offshore areas. While the CalCOFI record provides many examples of how marine communities in the SCCE respond to climate forcing, Koslow et al. (2018) caution that the impacts of secular climate change are unlikely to manifest as simply an extension of the warm phase of the ENSO or PDO cycles, and therefore past trends may not always be reliable guides for future ecosystem responses, which emphasizes the need for continued monitoring.

Monitoring programs will be essential to track the progression of anthropogenic climate change and understand associated ecosystem impacts. We suggest that existing oceanographic monitoring programs conduct an internal review to assess how the data collected from these programs can better contribute to stakeholder needs related to climate change adaptation. For countries working on developing new oceanographic monitoring programs to address climate issues, the following elements of CalCOFI have contributed in a valuable way to understanding the effects of climate change on the marine ecosystem: sampling inshore to offshore has revealed that nearshore trends are more pronounced, sampling below the thermocline in the upper 500 m of the water column has shown different manifestations of climate forcing, the collection of paired biological and hydrographic data allows for analysis of trends between environmental change and ecological responses, the positioning of the CalCOFI grid in a biogeographic boundary zone means that ecological communities are more responsive to climate change, and sampling across multiple trophic levels allows for an understanding of ecosystem functioning and food web impacts of climate change. The biological sampling strategy of CalCOFI is powerful in that early life history stages of many species are captured, allowing for a comparison of community trends through time for both fished and unfished species. However, the methodology also provides challenges in inferring adult habitat distributions, adult fitness, and recruitment success, which require complementary additional surveys to address.

Given the wealth of research and data along the US West Coast, this region is well primed to develop and test innovative strategies for climate change adaptation for marine ecosystem management and the CalCOFI Program is a key part of this process.

DATA AVAILABILITY STATEMENT

The datasets analyzed for this study can be found in the CalCOFI Database for hydrographic data (http://new.data.calcofi.org) and the ERDDAP database for ichthyoplankton data (https://oceanview.pfeg.noaa.gov/erddap/search/index.html?page=1&itemsPerPage=1000&searchFor=CalCOFI).

AUTHOR CONTRIBUTIONS

NG and BS conceptualized the manuscript. NG performed the analysis and wrote the manuscript. ED contributed to the climate projection analysis. All authors contributed to the manuscript revision, and read and approved the submitted version.

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Marine Vertebrate Biodiversity and Distribution Within the Central California Current Using Environmental DNA (eDNA) Metabarcoding and Ecosystem Surveys

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Environmental DNA (eDNA) metabarcoding is a new approach for assessing marine biodiversity that may overcome challenges of traditional monitoring and complement both existing surveys and biodiversity assessments. There are limited eDNA studies that evaluate vertebrate biodiversity in the marine environment or compare patterns of biodiversity with traditional methods. This study uses eDNA metabarcoding of the mitochondrial 12S rRNA genes present in seawater samples to characterize vertebrate biodiversity and distribution within National Marine Sanctuaries located in the California Current upwelling ecosystem. The epipelagic community in the study region has been monitored using traditional (mid-water trawl and marine mammal) survey methods since 1983. During 2016 and 2017, we concurrently sampled the epipelagic community using traditional survey methods and water for eDNA analysis to assess agreement among the methods. We collected replicate eDNA samples from 25 stations at depths of 10, 40, and 80 m, resulting in 131 small volume (1 L) environmental water samples to examine eDNA sequences. Across the eDNA and traditional survey methods, 80 taxa were identified. Taxa identified by eDNA partially overlapped with taxa through trawl and marine mammal surveys, but more taxa were identified by eDNA. Diversity and distribution patterns of marine vertebrates inferred from eDNA sequences reflected known spatial distribution patterns in species occurrence and community structure (e.g., cross-shelf and alongshore patterns). During both years, we identified fishery taxa *Sebastes* (rockfish), *Merluccius* (hake), *Citharichthys* (sanddab), and *Engraulis* (anchovy) across the majority of the stations using eDNA metabarcoding. The marine vertebrate assemblage identified by eDNA in 2016 was statistically different from the 2017 assemblage and more marine mammals were identified in 2017 than in

2016. Differences in assemblages identified by eDNA were coincident with different oceanographic conditions (e.g., upwelling and stratification). In 2016, weak upwelling and warmer than average conditions were measured, and vertebrate assemblages were not different among ecological regions [Point Reyes, Pescadero, and Monterey Bay]. While in 2017, average upwelling conditions returned, vertebrate assemblages differed at each region. This study illustrates that eDNA provides a new baseline for vertebrate assessments that can both augment traditional biomonitoring surveys and aid our understanding of changes in biodiversity.

Keywords: environmental DNA, marine biodiversity, vertebrates, fish, marine mammals, biomonitoring, ecosystem oceanography, marine sanctuaries

INTRODUCTION

Marine biodiversity is in decline globally in part due to overfishing, pollution, and climate change (Jackson et al., 2001; Pauly et al., 2002; Cheung et al., 2013; Barange et al., 2014; Haigh et al., 2015; McCauley et al., 2015; Somero et al., 2016). Due to the vastness and inaccessibility of the ocean, biodiversity patterns in pelagic ecosystems are difficult to assess (National Research Council, 1992; Kaschner et al., 2006). As a result, efforts to monitor and detect changes in biodiversity are limited. Effective conservation policies to protect biodiversity are critical to maintaining healthy and resilient ecosystems (Sciberras et al., 2013; Lubchenco and Grorud-Colvert, 2015; O'Leary et al., 2016), yet are challenging to implement. Future efforts to manage and conserve marine resources would benefit from the evaluation and application of new biological monitoring (biomonitoring) technologies to improve capabilities to monitor species abundance, diversity, and distribution patterns in marine ecosystems.

Traditional biomonitoring of pelagic vertebrate distributions includes use of net trawl surveys to assess mid-water and benthic organisms, and visual surveys to assess air-breathing vertebrates (e.g., marine mammals) (Barlow and Forney, 2007; Keller et al., 2012; Sakuma et al., 2016). Although fisheries-dependent data (e.g., catch per unit effort from fishing vessels) have been used to inform fish population models since the early 20th century, fishery independent monitoring (from trawl or visual surveys) of marine fish and mammals has only been common practice since about the 1960s (Gunderson, 1993). Around the same time, government-led assessments of targeted fisheries began along with the collection of landings data as stocks began to decline (Edwards et al., 2010).

Off the coast of the Western United States, the California Current Ecosystem (CCE) is a productive eastern boundary upwelling system that stretches from the Strait of Juan de Fuca to Baja California along the Washington, Oregon, and California coasts. The CCE has high seasonal primary productivity that contributes to marine biodiversity that is of both economic and conservation importance (Checkley and Barth, 2009; Fautin et al., 2010). Off central California, the National Oceanic and Atmospheric Administration (NOAA) National Marine Fisheries Service conducts an annual Rockfish Recruitment and Ecosystem Assessment Survey (RREAS) during peak springtime upwelling conditions. Since 1983, this annual survey of juvenile rockfish and other groundfish has used trawl and acoustic survey methods to monitor the distribution, abundance, and biodiversity of pelagic micronektonic organisms, and visual surveys (e.g., line transect) to monitor the abundance and distribution of seabirds and marine mammals (Santora et al., 2012, 2017; Ralston et al., 2015; Sakuma et al., 2016). The RREAS represents one of the longest-running time series of epipelagic juvenile fishes and other micronekton (referred to here on as "forage assemblages") monitoring in the world (Edwards et al., 2010; McClatchie et al., 2014). Using data from the RREAS, researchers have assessed the environmental drivers of juvenile rockfish abundances, spatial ecology of forage assemblages and their relationships with seabird breeding performance and spatial distribution, as well as documented baseline patterns of pelagic biodiversity (Santora et al., 2012, 2014, 2017; Ralston et al., 2013; Schroeder et al., 2018). Stations from the RREAS adhere to regional zones, and long-term datasets have shown these regions to be ecologically important for particular species and distributions of those organisms can be spatially explicit to specific regions (Santora et al., 2012). The present study augments the RREAS by using environmental DNA (eDNA), the DNA shed by organisms into the environment, to biomonitor vertebrates concurrently.

eDNA can be isolated from water samples and analyzed to detect unique sequences from microorganisms to large vertebrates, thus allowing biodiversity assessments to be completed without visually observing organisms (Foote et al., 2012; Thomsen et al., 2012; Kelly et al., 2014; Djurhuus et al., 2017). eDNA has been used to indicate presence of invasive species (Pochon et al., 2013), assess changes in taxa assemblages over time (Sawaya et al., 2019), and track ecologically important marine species (Sassoubre et al., 2016). In the CCE, eDNA has been used to detect vertebrates in Monterey Bay (Port et al., 2016; Andruszkiewicz et al., 2017b) and off the coast of Santa Barbara (Lafferty et al., 2018). Other studies have compared marine vertebrate eDNA assessments to visual surveys (Thomsen et al., 2012, 2016; Port et al., 2016; Kelly et al., 2017; Yamamoto et al., 2017; Boussarie et al., 2018; Stat et al., 2019). While some of the aforementioned studies used DNA metabarcoding or quantitative polymerase chain reaction (qPCR) to investigate vertebrate biodiversity, our study expands on that body of work by examining eDNA metabarcoding of vertebrates in the CCE by comparing both concurrent net tows and marine mammal visual sightings to assess agreement among the methods.

This study compares oceanographic and biomonitoring data collected over two years within five geographic regions, spanning 539 km, including stations within the Cordell Bank, Gulf of the Farallones, and Monterey Bay National Marine Sanctuaries. We test hypotheses to examine whether vertebrate taxa identified as present through eDNA differ among the various regions and the two collection years. We compare oceanographic conditions within the two sampling years to correlate different environmental conditions with biodiversity patterns. eDNA biomonitoring data are compared to concurrently collected data on the distribution of epipelagic fish and marine mammals collected at the same stations and day to determine the level of taxa overlap between the three methods. We also discuss and compare eDNA results in relation to known pelagic biodiversity patterns and assess the utility of eDNA for monitoring the presence of difficult to survey marine vertebrates.

MATERIALS AND METHODS

Oceanographic Condition Surveys and Field Biological Collections

Biological samples and environmental data were collected from the NOAA Fisheries Survey Vessel *Reuben Lasker* during the 2016 and 2017 RREAS (NOAA project numbers RL-16-03 and RL-17-03). The cruises took place during April and May in both years. Sample collection occurred over six days in 2016 and five days in 2017. Each day, stations were sampled within a given region representing a subset of the full RREAS stations within the Cordell Bank, Gulf of the Farallones, and Monterey Bay National Marine Sanctuaries off the coast of California, United States (**Figures 1**, **2**; Sakuma et al., 2016). Concurrently, salinity and temperature were measured at each station using an SBE 9plus conductivity–temperature–depth (CTD) sensor (Sea-Bird Scientific, United States). Instrument casts were made to a depth of 500 m or to 10 m from the bottom in shallower locations, and measurements were obtained throughout the water column. To assess regional oceanographic conditions, we calculated the following at each station: depth-averaged mean sea temperature and salinity over 20–40 m (depth range chosen to match net haul depth), stratification strength as the integrated potential energy between 0 and 40 m (Ladd and Stabeno, 2012), and depth of the 26.0 isopycnal (Santora et al., 2014; Schroeder et al., 2014). The stratification strength and 26.0 isopycnal depth were calculated to assess vertical ocean conditions and to provide a relative index of upwelling conditions and nutricline depth. Further, to compare differences in oceanic conditions during 2016 and 2017 relative to past RREAS data, we calculated the spatial climatology of these variables over 1990–2017 (using April–May averages, Santora et al., 2014; Schroeder et al., 2014). These variables were spatially interpolated using the optimal interpolation scheme Divand[1] with a 15-km meridional correlation length and a 10-km zonal length. The data were interpolated onto a grid having a $0.1° \times 0.1°$ spacing, and the interpolation parameters were set to force the interpolations to match observations (**Figure 2**).

Epipelagic micronekton (free swimming organisms generally <200 mm) samples were collected at night using a modified Cobb mid-water trawl with a 9.5-mm cod-end liner, and 15-min tows were made at each station with a headrope depth of 30 m (Sakuma et al., 2016). Around 150 trawls are conducted each year during the RREAS. For this study, a total of 14 hauls (seven each year) were conducted at stations that overlapped with eDNA collections conducted earlier in the day. After each haul, organisms were separated, identified, and enumerated. Organism identification was either to the family, genus, or species level, depending on the degree to which the identity of the organism could be consistently discerned by survey staff at sea. Relative abundance was measured as catch-per-unit-effort (CPUE).

Visual marine mammal surveys were conducted concurrently during the 2016 eDNA water sampling, but not during the 2017 eDNA collection dates. Visual surveys for marine mammals were conducted during daylight hours while the vessel moved (at speeds greater than 5 knots) between hydrographic sampling stations (Santora et al., 2012). Visual surveys were conducted only during favorable sea-state conditions (e.g., no fog or glare and in Beaufort conditions <6). All sightings were recorded by an observer stationed on the flying bridge using binoculars and were entered into a mapping program that was synced to the ship's navigational system. Marine mammal sightings included line-transect methodology (e.g., distance and bearing of cetaceans) to a maximum distance of 2 km (Santora et al., 2012). All sightings data were stored in a relational database identified down to species level or coarser taxonomic levels and organized into 3-km bins. Sightings data for the five regions sampled in 2016 were subset and summarized.

Water samples for eDNA were collected on five days during the 2016 survey and four days during the 2017 survey. On each day, water samples were collected from three stations in a region during daytime CTD casts. The last cast of each day was just prior to the first mid-water trawl of the night. Seawater samples were collected from depths of 10, 40, or 80 m in triplicate (three Niskin bottles at each depth) with 10 L Niskins (General Oceanics, United States) on a 24-bottle rosette (**Supplementary Table S1**). It should be noted that it was operationally impossible to sterilize the Niskin bottles on the rosette sampler between sampling events. Using sterile 69-oz bags (Whirl-Pak; Nasco, United States), water samples were transferred from Niskin bottles to 250-ml single-use, sterile, Analytical Test Filter Funnels (Nalgene; Nunc, United States) fitted with 0.22-μm pore-size hydrophilic polyvinylidene fluoride (PVDF) filters (GVWP04700, Millipore, United States). Milli-Q ultrapure water (Millipore) was filtered to create a daily negative control collection blank sample (referred to by some authors as a filtration blank). A total volume of 1 L was filtered for both the environmental and collection blank samples with a six-valve vacuum manifold. A new sterile filtration funnel and filter were used for each sample. Filters were removed from funnels with sterile forceps and placed in sterile 5 ml screwcap tubes and frozen at −80°C until processing. Samples were labeled with the following naming convention: year_CTD number_station

[1] https://github.com/gher-ulg/DIVAnd.jl

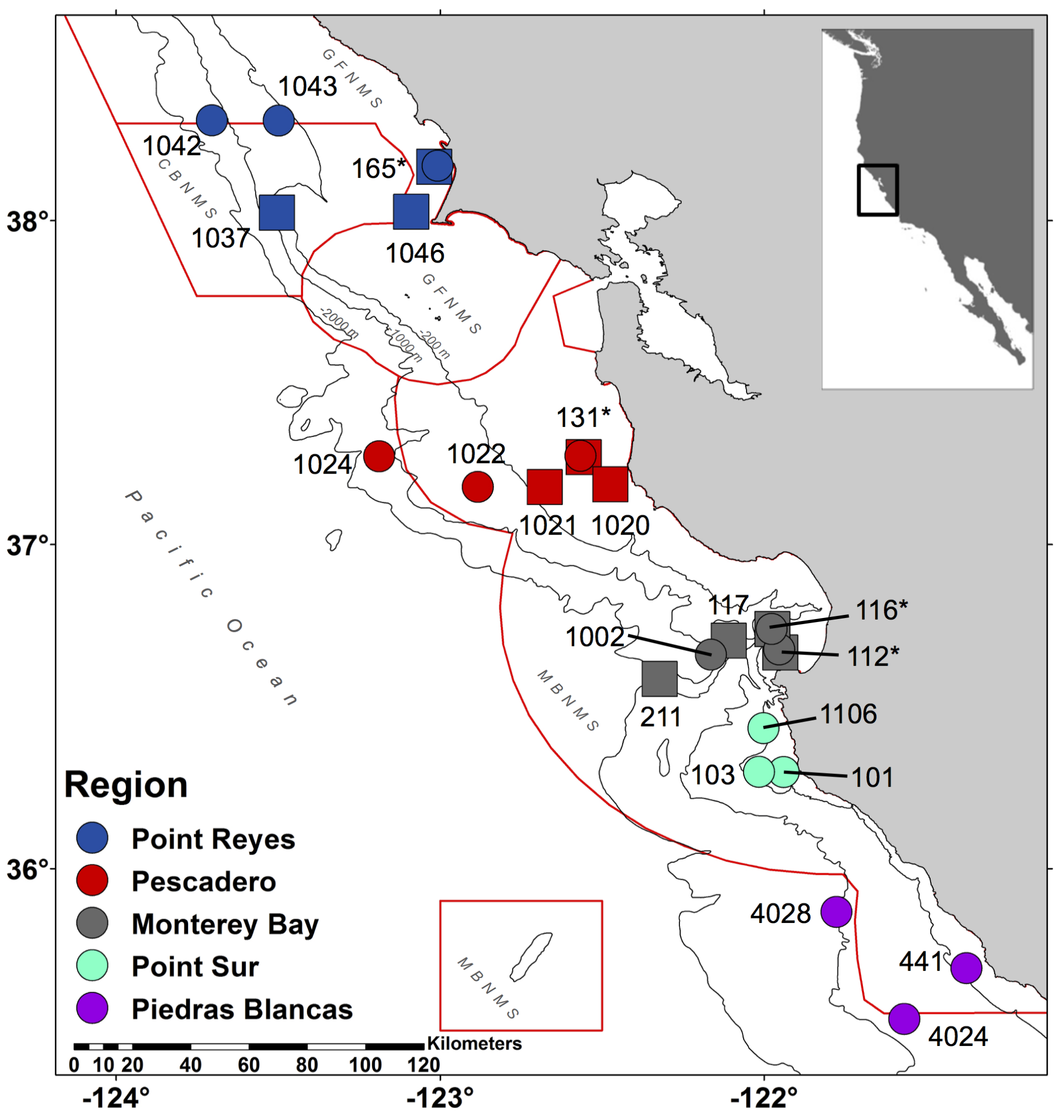

FIGURE 1 | Central California eDNA sampling map. This map indicates the stations where seawater was collected during 2016 (circles) and 2017 (squares). Each location, from north to south, is indicated by a color: Point Reyes (blue), Pescadero (red), Monterey Bay (gray), Point Sur (turquoise), and Piedras Blancas (violet). National Marine Sanctuaries are outlined in red from north to south: Gulf of the Farallones (GFNMS), Cordell Bank (CBNMS), GFNMS again, and Monterey Bay (MBNMS) with an additional red box around Davidson Seamount, which is part of MBNMS. Contours are 200, 1000, and 2000 m isobaths. Stations that were collected both years = *.

number_depth_replicate (e.g., 2016_006_116_80M_1). In order to process all filters at the same time and avoid laboratory bias, filters collected in 2016 were stored for 14 months prior to analysis while filters collected in 2017 were stored up to 2 months prior to analysis.

DNA Extraction

DNA was extracted from the filters using a modified DNeasy Blood & Tissue Kit (Qiagen, United States) extraction protocol with 0.75 g each of ashed and UV-irradiated 0.5-mm and 0.1-mm glass beads (BioSpec Products Inc., United States). For each day that extractions were performed, one extraction was conducted without a filter to serve as an extraction blank. In total, 188 samples were extracted: environmental samples (n = 164), collection blanks (n = 10), and extraction blanks (n = 14). Total DNA in each extract was quantified using a Qubit 2.0 Fluorometer and Qubit dsDNA HS assay (Invitrogen, United States) and then extracts were subsequently stored at −20°C (see **Supplementary Table S2**). The stepwise DNA extraction protocol is available at: dx.doi.org/10.17504/protocols.io.yrqfv5w. To serve as a positive control, an artificial community was constructed using DNA extracts from eight fish species (see **Supplementary Supporting Materials** and **Supplementary Table S3**). This artificial "mock" community is not meant to reflect the conditions encountered in seawater where DNA from fish is expected to be at much lower

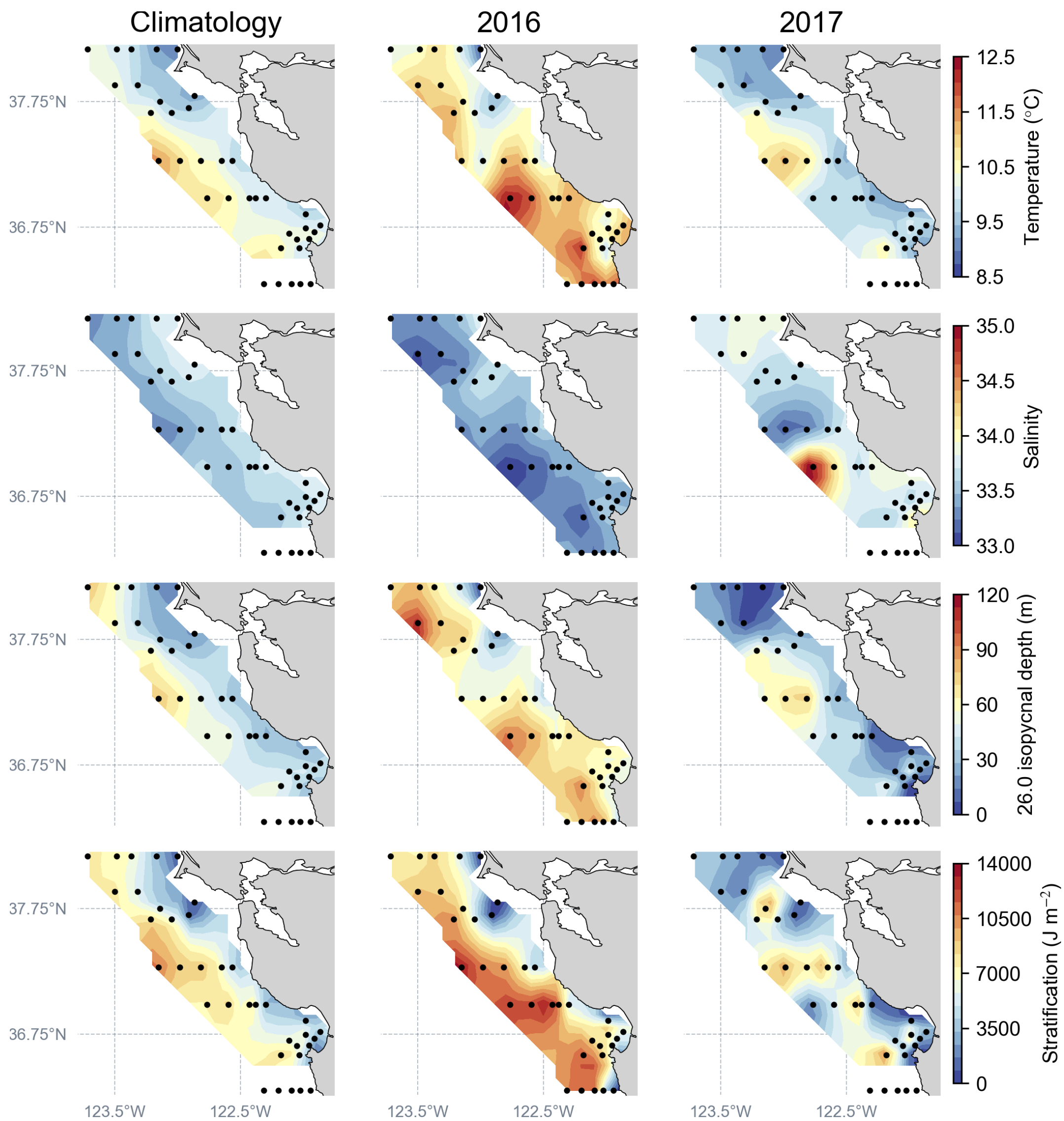

FIGURE 2 | Regional ocean conditions, determined by the full set of CTD sampling stations (black dots) during April–May, to construct the spatial mean climatology (1990–2017) compared to April–May 2016 and 2017. Ocean condition parameters plotted: temperature and salinity at 20–40 m depth, depth of the 26.0 isopycnal, and stratification strength.

concentrations. Artificial communities are increasingly being used in amplicon sequencing studies, and their use is considered among best practices (Pollock et al., 2018).

12S rRNA Gene Amplification and Sequencing

Amplicons were generated through a two-step PCR procedure (O'Donnell et al., 2016), with the first PCR using non-index primers and the second PCR using unique indexed primers. To amplify DNA in each sample, the mitochondrial 12S rRNA gene was amplified using the MiFish-U primers (Miya et al., 2015) (F: GTCGGTAAAACTCGTGCCAGC, R: CATAGTGGGGTATCTAATCCCAGTTTG), which were designed to amplify fish DNA. All PCR reactions consisted of 7.2 μl of PCR-grade water, 0.8 μl of MiFish-U forward and reverse primer (10 μM), 10 μl of Qiagen HotStarTaq Plus MMX (2X), and 2 μl of DNA template (diluted 1:10). Each sample (both 2016 and 2017 environmental samples, collection blanks, extraction blanks, and artificial community) was run in triplicate along with separate no template controls (NTCs) per group of triplicates. Thermal cycling conditions for the first PCR began with an initialization step of 95°C for 5 min followed by 40 cycles of denaturation at 95°C for 15 s, annealing at 55°C for 30 s, and extension at 72°C for 30 s, then held at 4°C. PCR products from the triplicate PCR reactions were then pooled, and amplification was confirmed through visualization in a 1.5% agarose gel stained with ethidium bromide. Amplified samples were retained for the second PCR so long as the corresponding NTC was confirmed to contain no visible amplicon and there was a band in the gel indicating presence of the PCR product (ca. 170 bp in length). All amplicons and NTCs (even when NTCs did not contain amplification products that were visible in the gel) were cleaned with the Agencourt AMPure XP (Beckman Coulter, United States) using the manufacturer's protocol and were used as template in the second PCR. The second PCR used the MiFish-U primers as stated above with the addition of a 6-base index (**Supplementary Table S2** provides index for each sample). A unique index was applied to the 5′-end of both the forward and reverse primers for each sample and NTC. The thermal parameters of the second PCR were 95°C for 5 min followed by 20 cycles at 95°C for 15 s, 57°C for 30 s, and 72°C for 30 s, then held at 4°C. After the second amplification, tagged triplicate PCR products were pooled, and both NTCs and pooled products were visualized in a 1.5% agarose gel stained with ethidium bromide. If NTCs contained no visible amplicon, the corresponding amplified samples (product visualized in the gel) were cleaned as described above. Cleaned NTCs from the first PCR were divided into three pools per year, then carried forward to the second PCR, and tagged with unique indices to create six pools for sequencing. DNA in the cleaned PCR products was quantified using a Qubit 2.0 Fluorometer and Qubit dsDNA HS assay.

Three sequencing libraries were prepared. Tagged samples were assigned randomly to one of the three sequencing libraries. To each library, 50 ng of DNA was added from both the environmental and artificial community samples. The volume of extract of collection blanks, extraction blanks, and NTC pools added was chosen based on the average volume of environmental samples added to each library (Library 1 = 2.12 μl, Library 2 = 1.93 μl, and Library 3 = 1.84 μl). Library pool concentrations were confirmed using a Qubit dsDNA HS assay. Libraries were prepared according to the manufacturer's protocols using 250 ng of DNA from each pool with the KAPA HyperPrep Kit (KAPA Biosystems, United States); each library was ligated with a NEXTflex DNA barcode (BIOO Scientific, United States) and cleaned with Agencourt AMPure XP. Each library was brought to 10 nM and run on a Bioanalyzer 2100 with a High Sensitivity DNA Assay (Agilent Technologies, United States) at the Stanford Functional Genomics Facility (SFGF) to confirm the library concentrations. All three libraries were sequenced at SFGF on an Illumina MiSeq platform using 250-bp, paired-end sequencing with the MiSeq Reagent kit V2 and 20% PhiX174 spike-in control.

Sequence Processing

Data obtained from the three libraries were processed using modified versions of both the banzai pipeline (O'Donnell, 2015) and methods described in Djurhuus et al. (2017). Filtering parameters are detailed in **Supplementary Table S4**. Paired forward and reverse reads were assembled with PEAR v0.9.6 (Zhang et al., 2014). Merged reads were then quality filtered with VSEARCH v1.8.0 (Rognes et al., 2016). Reads that passed quality filtering were demultiplexed and singletons were removed with the *awk* command. Primer sequences were removed from the reads using cutadapt v1.8.3 (Martin, 2011). Identical sequences were consolidated with a custom dereplication python script. Sequences were clustered using Swarm v2 (Mahé et al., 2015) to create operational taxonomic units (OTU; clustering = 97%). OTU sequences were annotated against the BLAST nucleotide database (BLASTN) (≥97% ID, *E*-value < 10^{-20}, wordsize ≥30) and annotations were further refined with MEGAN v5.10.6 (Huson et al., 2016) (≥97% ID, *E*-value < 10^{-25}, LCA 70, minimum score 140). We note that additional databases are currently available, including the one created recently by Sato and colleagues (Sato et al., 2018).

Sequences annotated to phyla other than Chordata were removed from the dataset. Annotations to *Bos*, *Canis*, *Homo*, *Neovison*, *Ovis*, and *Sus* genera were subsequently removed to retain OTUs exclusively classified as marine vertebrates (**Supplementary Table S2**). Each environmental and artificial community sample was then rarefied to 44,676 reads and read counts were then converted to binary data with the "rrarefy" and "decostand" functions, respectively, in vegan with R package v2.4-4 (Oksanen et al., 2017; R Core Team, 2017). This number of reads (44,676) was chosen because >95% of the environmental samples contained reads equal to or larger than this. Five environmental samples (2016_009_112_10M_2, 2016_104_1106_80M_2, 2017_005_0112_40M_3, 2017_009_1020_10M_3, and 2017_021_1046_10M_1) did not meet the rarefying threshold and were removed from the subsequent analyses (see **Supplementary Table S2** for number of sequences). Annotations were assigned to the species level, where possible (n = 970 out of 1396 OTUs were assigned to the species level, 69% of the OTUs, which comprised 69 unique species, see **Supplementary Table S5**).

Statistical Analyses

To investigate whether the vertebrate assemblage differed among years, regions, or sample location (e.g., on or off the shelf and above or below the pycnocline), we conducted analysis of similarities (ANOSIM) with the Plymouth Routines in Multivariate Ecological Research software package v6.1.18 (PRIMER 6) (Clarke and Gorley, 2015). In some cases, nMDS plots were created to visualize the multi-dimensional data.

The null hypotheses tested were as follows: (1) marine vertebrate assemblages (identified by trawl or eDNA) are the same in 2016 and 2017, (2) marine vertebrate assemblages (identified by trawl or eDNA) are the same in the different geographic regions during a specific year, (3) marine vertebrate assemblages (identified by trawl or eDNA) are the same regardless of depth of the water column where the sample was collected (on the shelf <100 m deep or off the shelf >100 m deep), and (4) marine vertebrate assemblages (identified by trawl or eDNA) are the same whether sample depth was above or below the pycnocline (defined herein as the depth where the Brunt-Välsälä frequency is maximum). When hypotheses focused exclusively on data from a single year, samples from all regions studied in that particular year were used to test hypotheses. We considered five different regions in the analysis [Point Reyes, Pescadero, Monterey Bay, Point Sur, and Piedras Blancas] based on historical work in this region that suggests they can harbor distinct vertebrate assemblages (Santora et al., 2012). When hypotheses used data across the two years, we only used regions that were sampled in both years (Point Reyes, Pescadero, and Monterey Bay). ANOSIM analyses were conducted using Jaccard distance matrices. All analyses were conducted with a statistical significance threshold of $\alpha = 0.05$.

To test null hypotheses 1–4, data described down to the lowest resolution for each method was used for the analyses. eDNA assemblages were represented using binary (presence/absence) OTU-level data from eDNA samples; and trawl assemblages were represented using binary mid-water trawl data with identifications down to the species level when available.

Vertebrate Assemblage Characterization and Biomonitoring Methods Comparisons

To characterize the assemblages and to allow for direct comparisons between eDNA, trawl, and marine mammal survey collections, organism identifications were truncated to the genus level. Further, we have reduced confidence in the species-level annotations for some species complexes. For example, two *Sebastes* species were added in equal DNA concentrations to the artificial community, but only one species was resolved in the annotations and the majority of the annotations were to the genus level, *Sebastes* (**Supplementary Table S6**). After truncating the species-level annotations, 95% of eDNA OTUs were annotated to the genus level with the remaining 53 OTUs to the family level, and 16 OTUs (two clades Ovalentaria and Eupercaria) that could not be resolved to the family level owing to disagreement among taxonomists on their family membership (Betancur-R et al., 2017).

When describing results from the different methods below, the organisms identified in the eDNA, trawl, or marine mammal visual survey data assigned to a genus, family, or clade are referred to as taxa. Taxa counts were converted to binary data (1 = presence, 0 = absence) for the comparisons. To illustrate the taxa diversity detected, we constructed a phylogentic tree using phyloT (Phylogenetic Tree Generator) and visualized the tree with the Interactive Tree of Life (Letunic and Bork, 2016). The tree can be visualized at https://itol.embl.de/tree/7393202124298011550441244. Additionally, we created an eDNA and trawl taxa accumulation curve using the "specaccum" function with the vegan package in R (Oksanen et al., 2017; R Core Team, 2017).

RESULTS

Regional Ocean Conditions

By comparison to the long-term spatial climatology, ocean conditions during 2016 indicate increased sub-surface (30 m) warming, lower salinity, a deeper depth of the 26.0 isopycnal, and overall increased stratification strength, a pattern influenced by the lingering effects of the strong northeast Pacific marine heat wave (Di Lorenzo and Mantua, 2016; **Figure 2**). During 2017, ocean-climate conditions returned to long-term averaged conditions, and compared to 2016, measurements indicated cooler sub-surface temperatures and higher salinity values throughout the study region (**Figure 2**). Furthermore, during 2017, shallower 26.0 isopycnal depths and decreased stratification were observed throughout the region relative to 2016, with local maxima occurring off Point Reyes and Monterey Bay, indicating return to normal or average upwelling conditions (**Figure 2**).

eDNA Sequences

Paired-end sequencing resulted in a total of 41.6 M reads (Lib1 = 10,979,812 reads; Lib2 = 14,524,732 reads; Lib3 = 16,141,481 reads). Two of 10 field collection blanks contained 3 and 13 reads, respectively; the rest returned 0 reads. The 14 extraction blanks contained between 0 and 70 reads, with 5 of the 14 extraction blanks having reads with a mean of 6.7 reads. Two of the six pooled NTCs had reads, 2016_NTC_pool_2 = 10 and 2017_NTC_pool_2 = 346 reads. Given the low number of reads returned on these control samples, we concluded that there was minimal contamination and no adjustment was made for the reads detected in these controls. The vast majority of the reads from the control samples (91%) were annotated to *Homo sapiens*. We verified that there were no statistically significant differences in assemblages (as measured by ANOSIM) between libraries (data not shown).

The three artificial community replicates contained 123,077, 180,446, and 187,899 reads. All genera that composed the artificial community were identified in each replicate; however, the proportion of sequences were not equally distributed across the genera (**Supplementary Table S6**). We therefore concluded that metabarcoding data from the environmental samples should not be interpreted quantitatively and instead should be used to infer the presence or absence of an OTU. The 131 environmental samples retained after rarifying initially contained between 44,676 and 273,795 reads per sample. The rarefied environmental samples consisted of 1,396 unique OTUs.

Marine Vertebrate Taxa Assemblage Identified by eDNA

The 1,396 unique OTUs identified in the environmental samples using eDNA metabarcoding were annotated to 65 distinct marine vertebrate taxa (including fish and mammals) where a taxon is defined as a unique genus level or coarser taxonomic assignment (**Figure 3**). Of the 65 marine vertebrate taxa detected, 26 were identified in both years, while 16 were observed only in 2016 and 23 were observed only in 2017 (**Table 1**).

A total of 54 fish taxa were identified, 36 taxa in 2016 and 38 taxa in 2017, with 20 of those taxa identified in both years. The fish taxa identified were from three classes: Actinopteri, Chondrichthyes, and Cyclostomata. The majority of taxa detected in both years (49 of 54) were bony fishes in the class Actinopteri. Chondrichthyes (sharks and rays) were also identified in both years, while Cyclostomata (jawless fish) were detected in 2017 only. The ubiquitous fish taxa, defined here as taxa identified at more than half of the stations sampled in 2016 or 2017, were *Sebastes* (rockfish), *Merluccius* (hake), *Engraulis* (anchovy), and *Citharichthys* (sanddab). *Stenobrachius*

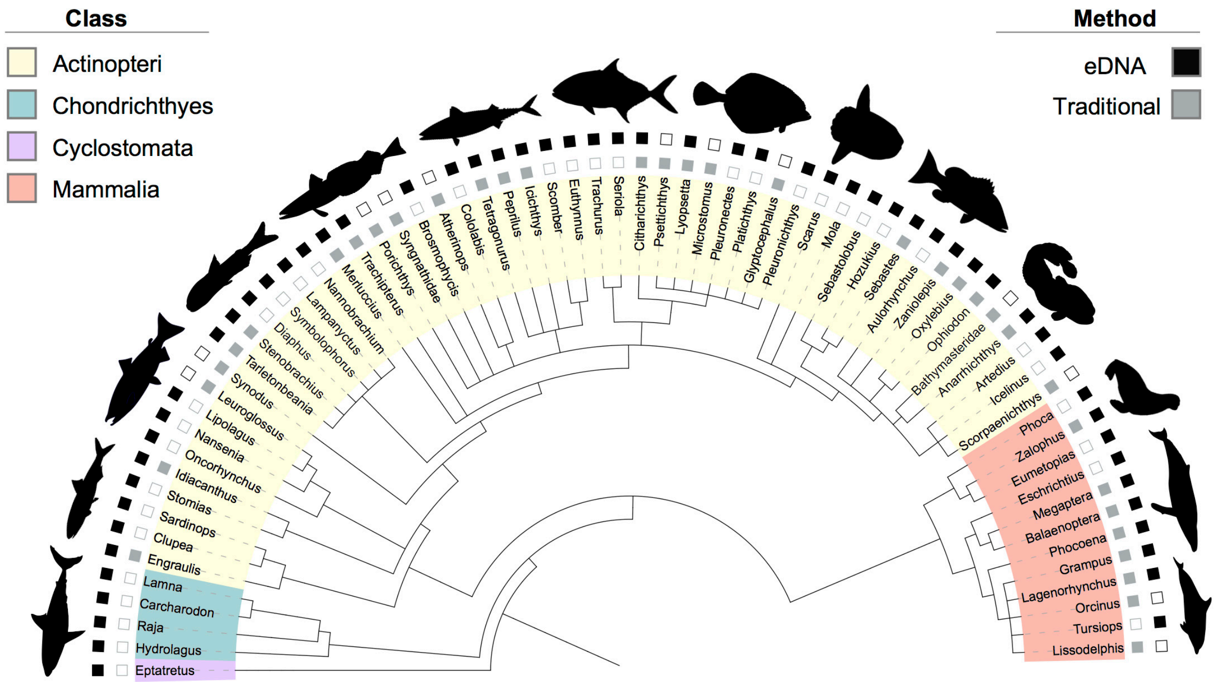

FIGURE 3 | Phylogenetic tree of the taxa diversity identified by eDNA, trawl, and marine mammal surveys across the two-year dataset. Taxa are annotated down to the most resolved level (genus). Classes are identified by leaf color. If a taxon was identified by eDNA, the corresponding outer square is solid (black); if identified by traditional methods (i.e., mid-water trawl or marine mammal visual surveys), the inner square is solid (gray). Empty squares denote no detection for that method. Bathymasteridae and Syngnathidae had no genera detected; therefore, the taxa family names are listed in the tree. Refer to **Table 1** for further information on which method detected certain taxa in a given year or region.

(lanternfish) was one of six Myctophidae genera identified in 2016 and was identified at the majority of stations during 2016 (**Supplementary Table S7**), while *Stenobrachius* was the only Myctophidae genera identified in 2017 but at less than half the stations.

A total of 11 marine mammal taxa, including Balaenopteridae and Eschrichtiidae (baleen whales), Delphinidae (dolphins), Phocoenidae (porpoises), Phocidae (seals), and Otariidae (sealions), were identified using using eDNA across the two years. Six marine mammal taxa were identified in 2016, all of which were also identified in 2017 along with five additional marine mammal taxa, totaling 11 identified marine mammal taxa in 2017. In 2016, Delphinidae were detected at the majority of stations (9/15, 60%).

Interannual and Regional Variability of Vertebrate Assemblage Identified by eDNA

Comparison of the five regions sampled in 2016 indicate that the vertebrate marine assemblage differed significantly across the regions ($R = 0.096$, $p = 0.002$). *Post hoc* pairwise comparisons for 2016 indicated that Point Sur differed significantly from the northern regions: Point Reyes ($R = 0.258$, $p = 0.001$), Pescadero ($R = 0.127$, $p = 0.048$), and Monterey Bay ($R = 0.198$, $p = 0.004$). The 2016 assemblage from the southernmost (Piedras Blancas) region also significantly differed from the northernmost region (Point Reyes) ($R = 0.165$, $p = 0.001$). Across these five regions, the 2016 assemblages identified on or off the shelf and collected above or below the pycnocline were not different ($p = 0.110$ and 0.219, respectively).

Three of the five geographic regions were sampled in both 2016 and 2017: Point Reyes, Pescadero, and Monterey Bay. Comparing eDNA from these three regions across years indicate that the 2016 marine vertebrate assemblage differed significantly from the 2017 assemblage ($R = 0.137$, $p = 0.001$). A closer look at these three regions in 2016 alone indicates no statistically significant difference in assemblage among regions ($R = 0.02$, $p = 0.261$), but in 2017, the eDNA assemblage varied significantly across the three regions ($R = 0.189$, $p = 0.001$; *post hoc* indicated all were different from each other, $p < 0.05$ for all pairwise comparisons), indicating increased regional heterogeneity in the assemblage. Using eDNA data from these three regions, we found that in both 2016 and 2017, samples collected on the shelf (<100 m) were significantly different from samples collected off the shelf ($R = 0.129$, $p = 0.046$; $R = 0.097$, $p = 0.007$, respectively, for the two years). The 2016 and 2017 vertebrate assemblages inferred from eDNA collected from water sampled above the pycnocline were not significantly different from water sampled below the pycnocline (2016: $p = 0.464$; 2017: $p = 0.061$). Although statistically significant, the R-values are small, suggesting that other factors affect the structuring of the community compared to those investigated herein. nMDS plots

TABLE 1 | Taxa identified across Point Reyes (PR), Pescadero (P), Monterey Bay (MB), Point Sur (PS), and Piedras Blancas (PB) in 2016 (circles), and 2017 (squares).

2016					Common Name (Taxon)	2017		
PB	**PS**	**MB**	**P**	**PR**		**PR**	**P**	**MB**
					Deep-sea smelt (Bathylagidae)^			□
		○			Deep-sea smelt, smoothtongue (*Leuroglossus*)			□
●		●	●	●	Deep-sea smelt, eared blacksmelt (*Lipolagus*)^	■		■
		●		●	Pencil smelt, Argentine (*Nansenia*)			
					Topsmelt, silverside (*Atherinops*)	□		
		○			Lizardfish (*Synodus*)			
					Midshipman (*Porichthys*)		□	
●	●		●		Saury, Pacific (*Cololabis*)	■		
				●	Jack, yellowtail amberjack (*Seriola*)‡	■		■
	●	●	●		Jack, Pacific jack mackerel (*Trachurus*)		■	■
				●	Herring, Pacific (*Clupea*)	■	■	■
		●		●	Sardine, Pacific (*Sardinops*)			■
◉	◉	◉	◉	◉	**Anchovy, northern (*Engraulis*)**	▣	■	■
					(Eupercaria)		■	
	●	◉	●	◉	**Hake, Pacific (*Merluccius*)**	■	▣	▣
	●				Parrotfish (*Scarus*)‡			
					Ribbonfish, King-of-the-salmon (*Trachipterus*)		■	▣
●					Lanternfishes (Myctophidae)			
		●	●		Lanternfish, California headlightfish (*Diaphus*)			
●		●			Lanternfish (*Lampanyctus*)			
●					Lanternfish (*Nannobrachium*)			
●	●	◉	●	●	**Lanternfish, northern lampfish (*Stenobrachius*)**		■	▣
●		●			Lanternfish, bigfin (*Symbolophorus*)			
●		●			Lanternfish, blue (*Tarletonbeania*)			□
●			●		(Ctenosquamata)			
●	●				Red brotula (*Brosmophycis*)	■		
					(Ovalentaria)	■		
					Wolf eel (*Anarrhichthys*)	■		□
	●				Tube-snout (*Aulorhynchus*)			■
					Ronquil (Bathymasteridae)		□	
					Sculpin (*Artedius*)	■		
					Sculpin (*Icelinus*)	■		
●	◉	●	●		Lingcod (*Ophiodon*)	□	□	▣
					Greenling, painted (*Oxylebius*)	■		□
					Rockfishes (Sebastidae)	■		
	●				Rockfish, NW Pacific (*Hozukius*)‡			
◉	◉	◉	◉	●	**Rockfishes (*Sebastes*)**	■	▣	▣
●	●		●		Rockfish, deep sea (*Sebastolobus*)			■
			○		Combfish (*Zaniolepis*)	■		■
					(Zoarcales)	■		
◉	●	◉	◉	◉	**Sanddab (*Citharichthys*)**	■	▣	▣
			◉	◉	Flatfishes, righteye (Pleuronectidae)	▣	▣	▣
		○	○		Flatfish, Rex sole (*Glyptocephalus*)			□
		●	◉		Flatfish, slender sole (*Lyopsetta*)		▣	
					Flatfish, Dover sole (*Microstomus*)			□
					Flatfish, starry flounder (*Platichthys*)	■		
				●	Flatfish, plaice (*Pleuronectes*)‡	■	■	
					Flatfish, turbot (*Pleuronichthys*)	■		■
					Flatfish, sand sole (*Psettichthys*)		□	□
		●			Salmon (*Oncorhynchus*)		■	■

(Continued)

TABLE 1 | Continued

2016					Common Name (Taxon)	2017		
PB	PS	MB	P	PR		PR	P	MB
○		●	◉	●	Medusafish (*Icichthys*)			
					Tuna, black skipjack (*Euthynnus*)‡			■
					Mackerel, Pacific (*Scomber*)		■	
			○		Pompano, Pacific (*Peprilus*)		■	■
		●		●	Squaretail (*Tetragonurus*)			□
○	○				Cabezon (*Scorpaenichthys*)			□
	●				Blackdragon (*Idiacanthus*)			□
●		●			Dragonfish (*Stomias*)			
	○				Pipefish (Syngnathidae)			
		●	●	●	Mola, common ocean sunfish (*Mola*)			■
					Ratfish, spotted (*Hydrolagus*)			■
	●			●	Shark, white (*Carcharodon*)			
●					Shark, salmon (*Lamna*)			
					Skate (*Raja*)	■		
					Hagfish (*Eptatretus*)	■		
					Sea lions (Otariidae)*^			
					Sea lion, Steller (*Eumetopias*)	■		
	●				Sea lion, California (*Zalophus*)*	■	■	
					Seal, harbor (*Phoca*)	■		
					Whales, rorquals (Balaenopteridae)*^			
					Whales, baleen (*Balaenoptera*)*			■
●					Whale, humpback (*Megaptera*)*			■
●	●	●	●	●	**Dolphins, oceanic (Delphinidae)**	■	■	■
		●			Dolphin, Risso's (*Grampus*)*	■		
	●				Dolphin, Pacific white-sided (*Lagenorhynchus*)*			■
					Dolphin, northern right whale (*Lissodelphis*)*			
					Orca, killer whale (*Orcinus*)*			
●	●	●		●	Dolphin, bottlenose (*Tursiops*)			■
					Whale, gray (*Eschrichtius*)	■		■
					Porpoise, harbor (*Phocoena*)*	■		

Identified by eDNA (solid), by trawl (empty), or both (haloed). Taxa are annotated down to the family level or clade level when no genus or family was resolved. See ***Supplementary Figure S2*** *for 2016 marine mammal visual survey transect observations. eDNA ubiquitous taxa = bold, identified by marine mammal survey in 2016 = *, improbable taxa = ‡, taxa not resolved beyond family by visual = ^.*

in **Supplementary Figure S1** illustrate weak clustering visually for statistically significant associations.

Marine Fish Assemblage Identified by Mid-Water Trawl

From the 14 mid-water trawls conducted (seven trawls each year), 28 unique vertebrate taxa were identified. In 2016, 16 taxa were identified by trawl, and in 2017, 23 taxa were identified. Eleven of the 28 trawl taxa were identified in both years. Trawl fish taxa in hauls corresponding to eDNA sampling were made up entirely of bony fishes, Actinopteri. The orders Perciformes and Pleuronectiformes (ray-finned and ray-finned demersal fishes) comprised the largest proportions of the taxa identified by trawl (6 out of 28, 21% each, 42% combined). Most of the taxa identified by trawl are bony fishes at the juvenile life stage that eventually grow beyond lengths of 200 mm. There were five ubiquitous fish taxa (defined previously) identified in the trawl samples; three taxa (*Citharichthys*, *Engraulis*, and *Sebastes*) and four taxa (*Citharichthys*, *Merluccius*, *Ophiodon*, and *Sebastes*) in 2016 and 2017, respectively.

Interannual and Regional Variability of Fish From Mid-Water Trawl Surveys

In 2016, the fish assemblage significantly differed among the five sampled regions ($R = 0.868$, $p = 0.029$). Most regions did not have replicate trawl collections that overlapped with eDNA collections, so *post hoc* comparisons could not be completed. Fish assemblage was not significantly different at stations on or off the shelf, and where trawls were conducted above or below the pycnocline ($p = 0.714$ and 0.857, respectively).

Comparing the fish assemblages observed in 2016 with those observed in 2017, there was no significant difference in the fish assemblages among the three regions that were sampled both years regardless of whether the data were examined in aggregate or by year (Point Reyes, Pescadero, and Monterey Bay; $p > 0.05$ for all). There was no significant difference between fish assemblages collected on-shelf vs. off-shelf in 2016 ($p = 0.714$) or in 2017 ($p = 0.286$). Further, fish assemblages collected in trawls conducted above and below the pycnocline were not significantly different in 2016 ($p = 0.99$). In 2017, trawls occurred exclusively below the pycnocline; thus, no pycnocline comparisons could be conducted.

eDNA Compared to Mid-Water Trawl Fish Survey

Combining both years, 48 fish taxa were identified by eDNA, of which 17 (35%) were also identified by trawl. An additional six taxa were identified by eDNA, but were considered implausible identifications due to either (1) mismatches between the observed and expected distribution of the taxa or (2) where genus-level identification cannot be easily discerned by visual identification (see **Table 1**). Thus, those six taxa are not included in the comparison of taxa identified by trawl and eDNA that follows. In total, 28 fish taxa were identified by trawl over the 2 years, of which 11 (39%) were identified by trawl and not by eDNA. Bathylagidae and *Leuroglossus* (both deep-sea smelts), *Atherinops* (topsmelt), *Synodus* (lizardfish), *Porichthys* (midshipman), Bathymasteridae (ronquils), *Scorpaenichthys* (cabezon), Syngnathidae (pipefish) as well as multiple Pleuronectidae–*Glyptocephalus* (rex sole), *Microstomus* (Dover sole), and *Psettichthys* (Pacific sand sole)– were all identified exclusively by trawl.

In 2016, 9 fish taxa were identified by both eDNA and trawl, 23 additional taxa were identified by eDNA only, and 7 fish taxa were identified by trawl only. Of the three regions from 2016 that were common between the two years, 8 fish taxa were identified by both eDNA and trawl, 18 additional taxa were identified by eDNA only, and 5 fish taxa were identified by trawl only. In 2017, 12 fish taxa were identified by both eDNA and trawl, 23 additional taxa were identified by eDNA only, and 11 fish taxa were identified by trawl only (**Table 1**).

eDNA Compared to Marine Mammal Survey

During 2016, over six days, a total of 539 km were sampled using visual survey methods to map the distribution of marine mammals. Ten mammal taxa were identified by visual survey. Of those taxa, four (*Grampus*, *Lagenorhynchus*, *Megaptera*, and *Zalophus*) were also identified by eDNA. In addition, *Tursiops* (bottlenose dolphin) were identified by eDNA in 2016, but were not identified by visual survey. eDNA did not detect *Balaenoptera* (blue or fin whale), *Orcinus* (killer whale), or *Lissodelphis* (right-whale dolphin) in 2016, which were detected by visual survey. Balaenopteridae (baleen whales) and Otariidae (eared seals or sea lions) were also detected by visual survey; although genera within those families were detected by eDNA, they were identified to a more resolved taxonomic level (*Megaptera* and *Zalophus*, respectively) (**Table 1** and **Supplementary Figure S2**).

DISCUSSION

In this study, eDNA metabarcoding identified more fish and marine mammal taxa than visual observations. Most of the fish and marine mammal taxa we identified using eDNA metabarcoding have been observed by trawl and marine mammal surveys during the RREAS over the last 35 years. Recognizing that eDNA was collected at more stations than the compared trawls, cumulatively, the molecular (eDNA) and traditional (trawl and marine mammal survey) observations identified 80 vertebrate taxa across the two years. eDNA detected 65 (81%) of these taxa, with more than half of the vertebrate taxa ($n = 42$) detected exclusively by eDNA. Of the 80 taxa, 13 taxa (11 fish and 2 marine mammal taxa) were identified exclusively with visual surveys and were not detected with eDNA. In 2016, the marine mammal survey identified more marine mammal taxa than did eDNA. These differences in the taxa detected by the different methods are not surprising because the trawl is designed to sample micronekton (e.g., epipelagic fish and invertebrates at <200 mm) using specific net mesh at a particular depth. Additionally, marine mammal visual surveys are conditional on sea state and number of observers conducting the survey. There were no marine mammal surveys to compare with eDNA in 2017, but the number of marine mammal taxa identified by eDNA that year was greater than was observed by both eDNA and visual survey in 2016. Notably, *Eschrichtius* (gray whale), *Eumetopias* (Steller sea lion), and *Phoca* (harbor seal) were additional taxa identified by eDNA in 2017. However, most of the taxa identified were not novel taxa for the area surveyed. Most of the fish taxa detected by eDNA have historically been identified by the last 2,200 trawls conducted since 1983, which have identified 116 unique genera or family taxa. All of the marine mammal taxa identified by eDNA have previously been identified by RREAS visual surveys.

Most taxa identified by eDNA were detected at only a few stations, but ubiquitous taxa were detected across the majority of stations either year. Of the fish taxa identified by eDNA over the two years, five were ubiquitous; four [*Sebastes* (rockfish), *Merluccius* (hake), *Engraulis* (anchovy), and *Citharichthys* (sanddab)] of which are managed fisheries along the West Coast of the United States. The distribution of these taxa also corresponds to the regionalization of species assemblages and their association with coastal upwelling patterns (Santora et al., 2012). Further, these taxa are an important component of seabird and marine mammal diet within the CCE (Szoboszlai et al., 2015), and their distribution and abundance are linked to their reproduction and population dynamics (Santora et al., 2014; Wells et al., 2017a; Warzybok et al., 2018). Most marine mammal taxa were also only detected with eDNA at a few stations, but Delphinidae were detected at the majority of stations in 2016. Previous visual surveys suggest that the distribution of most marine mammal taxa indicate specific habitat associations (i.e., on-shelf vs. off-shelf), whereas some taxa, such as dolphins, are considerably more broadly distributed

(Santora et al., 2012). Therefore, the identification of broad and specific distribution patterns of taxa identified by eDNA compared to previous documented patterns from trawl and visual surveys lends credence to the use of eDNA metabarcoding for biomonitoring in this region.

Oceanic climate conditions differed substantially between 2016 and 2017, owing in part to the influence of the large marine heat wave and El Niño during 2015–2016 (Fiedler and Mantua, 2017). Upwelling was weak in 2016, as indicated by increased stratification and the depth of the 26.0 isopycnal, with relatively minimal mixing compared to the 27-year average for the region. By comparison to the long-term average, oceanographic conditions during 2017 are representative of an average upwelling year, with development of known upwelling shadows and retention zones in Gulf of the Farallones and northern Monterey Bay (Graham and Largier, 1997; Santora et al., 2012), as indicated by increased nearshore salinity, decreased stratification, and shoaling of the 26.0 isopycnal depth (**Figure 2**). Previous analysis of micronekton biodiversity derived from the trawl survey during the marine heat wave (2015) found increased species diversity throughout the study region, which differed markedly compared to a strong upwelling year (Santora et al., 2017). However, the fish assemblages derived from the limited number of trawls available during the eDNA sample collection period indicate that assemblages did not differ among regions (Point Reyes, Pescadero, and Monterey Bay) in 2016 or 2017, suggesting no discernable effect of the climatic conditions on the geographic distribution of fish assemblages. In contrast, the fish assemblages inferred from eDNA were not different among the regions in 2016, but were different among the regions in 2017. This may be suggestive of an effect of the ocean climate conditions on fish assemblages and that eDNA samples, due to their ability to sample a broader suite of taxa, were sufficient to detect a difference in species assemblages within an average upwelling year. Moreover, it is possible that there were not a sufficient number of trawl samples compared in this study to assess a difference in species assemblage distribution. For example, the long-term multivariate index of species assemblages for the trawl survey, based on all trawls within the survey area, indicate that species assemblages were at an average level during the return to normal upwelling conditions in 2017 following the marine heat wave (Wells et al., 2017b). Given that we contrasted two years involving unprecedented ocean warming and a normal upwelling year, the eDNA taxa identified and their distribution patterns help establish a baseline to develop further work to investigate the effect of interannual changes in oceanic regimes on fish populations using eDNA.

Based on the long-term average of the trawl survey, the primary mode of spatial variability for taxa assemblages reflects a clear separation of assemblages on-shelf vs. off-shelf (Santora et al., 2012). Taxa assemblages inferred from eDNA on the shelf were distinct from those off the shelf during both years. This difference between eDNA identified vertebrate assemblages on-shelf vs. off-shelf was also observed by Andruszkiewicz et al. (2017b) within Monterey Bay in 2015. This may reflect differences in preferred habitat of the taxa or their prey habitat preference. eDNA taxa assemblages above and below the pycnocline were not different. Many mesopelagic species exhibit diurnal vertical migration and therefore it is not surprising that no differences between assemblages above and below the pycnocline were detected by eDNA. Samples collected at a different time of year, from depths deeper than 80 m, or closer to the benthos, may provide contrasting assemblages from the epipelagic zone.

Geographic distributions differed between the years among some common taxa. For example, Myctophidae, a common mesopelagic fish family, was more diverse in 2016. Six different Myctophidae genera were identified by eDNA across the regions in 2016 compared to just one genus in 2017. The genus *Stenobrachius* was distributed across the majority of the 2016 stations, while in 2017, *Stenobrachius* was the only Myctophidae genera detected with eDNA and was detected at a minority of the stations (3 of 10). All of these identified Myctophidae taxa have previously been identified by the RREAS trawls within the studied region. Important fishery taxa such as *Engraulis* (anchovy) were identified by eDNA across all the regions both years, while *Sardinops* (sardine) were rarely detected. Anchovies were identified in every companion trawl from 2016, but in only one companion trawl in 2017, and sardines were not identified in the compared trawls from either year. Though sardines usually occur farther offshore than anchovies, the oscillating pattern between these two taxa has been historically noted (Rykaczewski and Checkley, 2008). While the population numbers of both fisheries are presently low compared to historic averages, directed sardine fishing has been closed since 2015 due to continued low population numbers (Thayer et al., 2017; Hill et al., 2018).

Several taxa that are not typically caught in the trawl survey, by virtue of their larger size and ability to avoid trawl gear, were identified using eDNA metabarcoding. For example, *Oncorhynchus* (salmon) was identified by eDNA in Monterey Bay in both 2016 and 2017 as well as near Pescadero in 2017. *Clupea* (herring) were detected with eDNA in Monterey Bay in 2017 and near Point Reyes in both 2016 and 2017, but are rarely caught by the trawl survey. Large chondrichthyans, such as *Carcharodon* (white shark), have low vulnerability to this trawl gear and thus rarely encountered in survey catches, but were identified using eDNA at Point Reyes both years. Similarly, *Hydrolagus* (ratfish) and *Lamna* (salmon shark) were identified by eDNA in 2016 and 2017, respectively, while *Raja* (skate) was identified in 2017 by eDNA. Some taxa identified by eDNA were extremely rare among the environments sampled and have known historical distributions, which do not typically overlap with the study's regions, making their presence in the sample highly implausible [e.g., *Euthynnus*, *Scarus*, and *Seriola* (tropical); *Hozukius* (Northwest Pacific); and *Pleuronectes* (Atlantic and Alaska)]. For some of these taxa, the targeted gene region may not discriminate these taxa from closely related taxa or the representative entry within GenBank may have been mis-annotated (Tripp et al., 2011; Iwasaki et al., 2013; Heller et al., 2018). Additional work should be conducted to improve primer resolution, and additional voucher sequences from verified specimens should be deposited in open-access databases to improve species annotations of conserved gene regions. Future studies should consider manually curated reference databases like MitoFish (Sato et al., 2018) to improve sequence annotations.

Although visual surveys and eDNA detection of marine mammals did not yield sufficient sample sizes for direct comparisons, the distribution patterns of sightings and eDNA-based detections provide substantial support for eDNA methods for detecting marine mammals. Both visual survey and eDNA found nearshore species such as harbor porpoise close to land, clusters of baleen whales within Monterey Bay, and offshore species, such as Pacific white-sided dolphins and Risso's dolphins, within outer slope habitat. The primers used for eDNA metabarcoding were optimized for fish (Miya et al., 2015); however, most of the organisms identified by the marine mammal visual survey were detected with eDNA. New mammal-specific primers targeting the mitochondrial 12S rRNA gene are now also available and can be used in future studies (Ushio et al., 2017). Marine mammal visual surveys were conducted during transects and not at station points; therefore, not all visual detections could correspond in space with eDNA collected at stations. Not all detections corresponded in time either, which could be due to individuals not being visually observed, multiple individuals from the same taxon in the study region, or movement of marine mammals. The mammal taxa identified by eDNA and visual surveys confirm known habitat associations of marine mammals within the study region (Santora et al., 2012).

One of the limitations to the study is that eDNA samples could not logistically be collected via rosette at the exact same time that trawl or visual surveys were conducted. The ship needs to move for trawls and mammal surveys are conducted while the ship transits between stations. Water samples must be collected while the ship is stationary, and to avoid trawl debris from contaminating eDNA samples, it was necessary to conduct water sampling prior to trawls. Overall, the eDNA sampling was matched to the time and location of the trawls as well as possible. The traditional surveys have not been designed to detect some of the organisms that are detected by eDNA (e.g., large bony fish), as some taxa may evade the trawl net or are difficult to assess via the marine mammal survey [e.g., beaked whales and small porpoises (Barlow and Forney, 2007)]. This study was conducted for an average of six days during both years, a subset of the RREAS's typical assessment of 30–40 days involving hundreds of trawls. During this evaluation period, differences were noted by eDNA that were not identified by trawl. This may be due to the statistical tests conducted that were run with data at different resolutions, where eDNA comparisons were conducted using OTU-level data and trawl comparisons were conducted using data identified down to the family, genus, or species level. The differences noted between biomonitoring methods are undoubtedly in part due to the fact that twice as many taxa were identified using eDNA compared to trawls. This is likely due to a molecular advantage, where eDNA casts a "larger and finer resolution net," compared to a specifically designed trawl net to sample micronekton.

Many trawls are required to achieve an accumulated assessment of species biodiversity of micronekton. An accumulation curve of trawl vertebrate taxa (**Supplementary Figure S3**) suggests that more trawls are needed beyond the subset of trawls that were used in this study to achieve taxa saturation. Santora et al. (2017) examined 26 years of trawl survey effort and an estimated ∼500 trawls are required to achieve a robust understanding of species diversity in the RREAS. An eDNA taxa accumulation curve (**Supplementary Figure S3**) suggests that taxa diversity across 2016 and 2017 begins to saturate after 120 samples are included. However, it is unclear how many eDNA samples would be needed to capture the same diversity observed in RREAS trawls across 26 years. As more eDNA samples are collected within this region, a power analysis can be undertaken to investigate how many eDNA samples are needed to document assemblage and diversity patterns. At the present time, the computational methods and sufficient information on within-sample and between-sample variability are lacking to carry out a power analysis for the ANOSIM method. Future efforts to develop such power analysis techniques could build off of those completed for PERMANOVA (Kelly et al., 2015) or Dirichlet Multinomial (La Rosa et al., 2012) methods (Knight et al., 2018). As described by Knight et al. (2018), power analysis remains a challenge in research involving amplicon sequencing data.

Traditional biomonitoring surveys also provide taxa abundance estimates, which this study does not address. Despite the noted differences among survey methods, eDNA offers novel insight as a tool to augment existing survey platforms, especially those aimed at monitoring biodiversity. However, as noted by other studies (Barange et al., 2009; Zwolinski et al., 2012), most fisheries surveys and species stock assessments are informed by relative abundance information, rather than presence–absence data, constraining potential applications of eDNA results in traditional fisheries management until reliable abundance information is achieved with molecular methods. Regardless, presence/absence information can still inform traditional surveys and assessments for some species, particularly uncommon marine mammals, large elasmobranchs, and other megafauna that are difficult to detect or resolve to a lower taxonomic classification with traditional survey methods. A novel application of presence-only eDNA biomonitoring may focus on assessment of indicator species that reflect extreme ocean climate and ecosystem shifts (e.g., sub-tropical fish species during an El Niño) to develop early warning signals to better inform the timing of expected shifts. Further, for some more traditional fisheries survey targets, it is well known that some populations, particularly those of coastal pelagic species, expand and/or contract their distribution and range in response to a combination of environmental factors and abundance levels (MacCall, 1990; Agostini et al., 2006), such that the distribution of a stock alone can provide insights into biomass levels (Barange et al., 2009). Similarly, some acoustic surveys require net sampling to apportion the species composition of the biomass estimates that are based on acoustic signals (Zwolinski et al., 2012). However, the net collections may not be conducted at the same time as the acoustic signal, trawl sampling limitations may constrain the number of trawls that can be collected, and the selectivity of different species to trawl gear may vary. Simultaneous eDNA collections could supplement, enhance, and help validate the species assignments of biomass in such surveys by providing presence information over considerably greater spatial scales. Furthermore, adding eDNA collection on acoustic-trawl surveys may improve our understanding of trophic interactions because eDNA could provide additional information on forage species assemblages and the occurrence of top predators (e.g., whales) that are interacting with both coastal pelagic and other forage species at local and regional scales (Fleming et al., 2016).

The comparison between eDNA and traditional biomonitoring methods highlights the novelty and strength of eDNA assessment for additional taxa compared to net tows and suggests that eDNA is a powerful tool for marine vertebrate detection. eDNA has enhanced the detection of organisms when paired with traditional biomonitoring methods (Kelly et al., 2017; Berry et al., 2019; Stat et al., 2019). Adding eDNA methods to the suite of biomonitoring techniques used during the RREAS and potentially other ecosystem biodiversity monitoring programs would enhance detection of organisms and aid in the biomonitoring of marine ecosystems. There is large uncertainty in how biodiversity responds to changes in climate, and increasing ocean-climate variability necessitates our better understanding of how marine vertebrates might respond. Baselines of biodiversity can be collected through eDNA, which can aid in the understanding of short-term or long-term changes by comparing to future collections (Jarman et al., 2018). While residence time for some species' eDNA have been reported (Sassoubre et al., 2016; Jo et al., 2019), more studies on the fate and transport of eDNA (Andruszkiewicz et al., 2017a, 2019; Collins et al., 2018) will improve the use of eDNA to better understand the diurnal, seasonal, or anomalous distribution of select organisms. Extending long-term biomonitoring programs to include eDNA could improve taxon detection and resolve long-term patterns or changes in species of concern (Berry et al., 2019). For commercially important and managed vertebrates where abundance may be desired, species-specific qPCR assays could be designed to target eDNA of a particular taxon (Sassoubre et al., 2016; Lafferty et al., 2018; Jo et al., 2019). While this study provides one example of eDNA assessment for pelagic ecosystem biomonitoring targeting fish and mammal biodiversity, the method can be expanded to also detect other groups such as seabirds (Ushio et al., 2018) and sea turtles (Kelly et al., 2014). In addition, eDNA surveys beyond vertebrates is possible (Kelly et al., 2017; Berry et al., 2019; Sawaya et al., 2019), making the biomonitoring of entire ecosystems by eDNA plausible. Future studies should consider connecting microorganism assemblages with invertebrate and vertebrate assemblages to have a more robust understanding of the biodiversity interactions within an ecosystem.

DATA AVAILABILITY STATEMENT

The raw sequence datasets generated for this study can be accessed under NCBI's BioProject PRJNA525068 and Sequence Read Archive (SRA) submissions SAMN11041033 to SAMN11041035.

AUTHOR CONTRIBUTIONS

CC, HS, EA, KS, JF, and AB conceived the study. CC, HS, EA, JS, and KS conducted the sample collection and lab analyses. CC, JS, IS, KS, JF, and AB conducted the data analysis. CC, JS, IS, and AB drafted the manuscript. All authors revised and edited 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.2019.00732/full#supplementary-material

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Gazing at the Crystal Ball: Predicting the Future of Marine Protected Areas Through Voluntary Commitments

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The beginning of 2015 saw a new era within the United Nations: the era of the sustainable development goals (SDGs). Built off the previous Millennium Development Goals, this new set of goals included 17 target areas, including, for the first time, an explicit global goal related to the ocean. In June 2017, at the United Nations Headquarters in New York City, a high-level conference surrounding SDG 14: Life Under Water convened. One dimension of goal 14 calls for 10% of the ocean conserved by the year 2020, through sub-target 14.5. That 10% fulfillment is often thought of in terms of areal coverage via marine protected areas (MPAs). While many objectives were laid out for this conference, one of the most prominent objectives was to build on existing partnerships and foster new collaborations. One way to achieve this target was through the creation of the voluntary commitment program. This "Call for Action" came from heads of state and government, as well as high-level representatives from organizations and stakeholder groups. Under this "Call for Action," 22 actions related to goal 14 were listed for stakeholders to partake in, including an appeal to create voluntary commitments surrounding the oceans. As of September 2017, 1,395 voluntary commitments had been registered through the voluntary commitment portal process, spanning across organizations and disciplines. Here, we analyze these commitments, specifically those related to the fifth sub-target of SDG 14. Commitments were further refined through spotlighting on those under 14.5 that focused on different forms of resilience. The resulting 133 separate codes covered over 12 distinct forms of resilience. Through analyzing commitments, we map out future plans and predict different forms of MPAs. This research shows collaboration and co-production of knowledge linking across the SDGs. This work can be seen as a stepping-stone to the fulfillment of 10% conservation by 2020.

Keywords: ocean, marine protected area, marine reserves, United Nations, sustainable development goals, resilience, voluntary commitments

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INTRODUCTION

Marine protected areas (MPAs) are regions of the ocean where specific human activities are limited or prohibited, and have been increasingly promoted by policy-makers, scientists, and conservationists as a tool for mitigating ocean threats, conserving biodiversity, managing fisheries, and enhancing resilience to climate change (Lester et al., 2009; Gaines et al., 2010;

Lubchenco and Grorud-Colvert, 2015; Roberts et al., 2017). In recent years, the global spatial extent of MPAs has increased across the world's oceans, with 4.8% of the world's ocean area currently under some form of protection in 2018 (MPAtlas.org, 2018).

The political calls for increased use of MPAs arise from numerous studies that demonstrate that MPAs – especially no-take MPAs (also known as marine reserves) – provide significant positive ecosystem benefits (Baskett and Barnett, 2015). These benefits include increases in biomass, density, size, and diversity of life in the region (Lester et al., 2009; Caselle et al., 2015). Benefits derived from MPAs also include benefits to fisheries, including by facilitating the recovery of depleted fisheries and by providing spillover effects (Gaines et al., 2010; Halpern et al., 2010). Further, because they maintain all trophic levels of the ecosystem and increase both species and genetic diversity, MPAs can enhance resilience to ecosystem changes, including those brought about by climate change (Olds et al., 2014; Roberts et al., 2017).

Despite an increasing trend toward implementing MPAs, doing so in international waters has proved to be a more difficult challenge. It was only recently that MPAs were created within what are commonly called "areas beyond national jurisdiction" (ABNJs) (Gjerde et al., 2008; Wells et al., 2016; Smith and Jabour, 2018). To date, only 1.2% of the high seas fall under protection[1], which comprises only 12 total MPAs within ABNJs governed by two different regional management bodies (Ardron et al., 2008; De Santo, 2018). Ten of these MPAs are under the management of the OSPAR Convention and the North East Atlantic Fisheries Commission. The remaining two are located in Antarctica and are managed under the Convention for the Conservation of Antarctic Marine Living Resources. Yet, ABNJs account for >60% of the global ocean by area and include critically important areas for biodiversity and ecosystem processes (Rogers et al., 2014; Gjerde et al., 2016).

The path to creating MPAs in ABNJs may become clearer. After 10 years of discussion, the United Nations General Assembly is adopting a resolution related to the sustainable use and conservation of the marine environment within ABNJs. This resolution will create an international legally binding instrument providing for the adoption of MPAs, as well as other key concerns such as marine genetic resources and environmental impact assessments (United Nations, 2017b). This process of United Nations meetings to create the legally binding instrument is set to end in 2020, although when the instrument will come into effect is still unknown.

Due to increasing loss of, and continued threats to marine biodiversity (McCauley et al., 2015) MPAs have become a focal point within international agreements and conferences in the last 10 years. This includes: the first International MPA Conference in 2005 and the adoption of the Convention on Biological Diversity (CBD) in 1992 (National Research Council, 2001). A number of international targets have been promulgated regarding the adoption of MPAs in national waters and in ABNJ. At the 2002 World Summit on Sustainable Development, participating States agreed to designate a global network of MPAs by 2012 encompassing 10% of all ecological regions (Gjerde et al., 2016). This call was further reiterated at both the 2003 and 2008 International Union for Conservation of Nature (IUCN) World Conservation Congresses, which called for protected areas to encompass 20–30% of all marine habitats (Gjerde et al., 2016). The 2010 Aichi Biodiversity Targets adopted by the Convention on Biological Diversity offered a new deadline of 2020 to designate 10% of the global ocean as protected areas. Finally, in 2014 the IUCN World Parks Congress recommended that 30% of the ocean be protected through the designation of MPAs. Given that <5% of the global ocean falls within MPAs, countries still have a long way to go to reach global targets and doing so in ABNJs will be a key element. Despite the challenges of creating MPAs in ABNJ, the increased development of MPAs globally has been accompanied by an interest in the role that MPAs can play in making marine systems more resilient to climate change impacts (Roberts et al., 2017). The increasing use of the climate resilience rationale for MPA creation (as opposed to a more traditional focus on biodiversity conservation) changes stakeholder perceptions of MPAs as policy instruments (Hopkins et al., 2016). Here, as we assess the future of MPAs in the new global ocean regime, the rise of the resilience rationale merits attention.

Sustainable Development Goal 14.5

In 2015, the United Nations agreed to 17 sustainable development goals (SDGs) to replace the previously held Millennium Development Goals. Under these 17 SDGs, goal 14 is often referred to as the Ocean goal, as its primary goal is "*to conserve and sustainably use the oceans, seas and marine resources for sustainable development*" (United Nations, 2017a). Goal 14 includes 10 subtargets relating to all things marine, such as ocean acidification and illegal, unregulated and unreported fishing (**Table 1**). With regards to MPAs, specifically, SDG 14.5 calls for that "[b]y 2020, [to] conserve at least 10 per cent of coastal and marine areas, consistent with national and international law and based on the best available scientific information." (United Nations, 2018). SDG 14.5 is measured in terms of success, which calls for "[c]overage of protected areas in relation to marine areas" (United Nations, 2018).

Sustainable development goal 14 is in its moment of prominence. In June 2017, a week-long, high-level United Nations conference met at the UN Headquarters in New York City to discuss the world's ocean, and specifically to advance the implementation of SDG 14. This meeting, called the *United Nations Ocean Conference*, was the first UN conference dedicated explicitly to discussing issues surrounding the marine environment. In addition to country delegations, participants included non-governmental organizations (NGOs), UN entities, academic institutions, civil-society organizations, inter-governmental organizations (IGOs), partnerships, as well as members of the private sector. Days were made up of partnership dialogs that focused on different ocean themes, as well as side events hosted by different state and non-state actors.

[1] mpatlas.org

TABLE 1 | SDG 14 subtargets and what thematic area they concern.

SDG 14 subtarget	Text of subtarget
14.1	By 2025, prevent and significantly reduce marine pollution of all kinds, in particular from land-based activities, including marine debris and nutrient pollution
14.2	By 2020, sustainably manage and protect marine and coastal ecosystems to avoid significant adverse impacts, including by strengthening their resilience, and take action for their restoration in order to achieve healthy and productive oceans
14.3	Minimize and address the impacts of ocean acidification, including through enhanced scientific cooperation at all levels
14.4	By 2020, effectively regulate harvesting and end overfishing, illegal, unreported and unregulated fishing, and destructive fishing practices and implement science-based management plans, in order to restore fish stocks in the shortest time feasible, at least to levels that can produce maximum sustainable yield as determined by their biological characteristics
14.5	By 2020, conserve at least 10% of coastal and marine areas, consistent with national and international law and based on the best available scientific information
14.6	By 2020, prohibit certain forms of fisheries subsidies which contribute to overcapacity and overfishing, eliminate subsidies that contribute to illegal, unreported and unregulated fishing, and refrain from introducing new such subsidies, recognizing that appropriate and effective special and differential treatment for developing and least developed countries should be an integral part of the World Trade Organization fisheries subsidies negotiation.
14.7	By 2030, increase the economic benefits to Small Island developing States and least developed countries from the sustainable use of marine resources, including through sustainable management of fisheries, aquaculture, and tourism
14.a	Increase scientific knowledge, develop research capacity, and transfer marine technology, taking into account the Intergovernmental Oceanographic Commission Criteria and Guidelines on the Transfer of Marine Technology, in order to improve ocean health and to enhance the contribution of marine biodiversity to the development of developing countries, in particular small island developing States and least developed countries
14.b	Provide access for small-scale artisanal fishers to marine resources and markets
14.c	Enhance the conservation and sustainable use of oceans and their resources by implementing international law as reflected in UNCLOS, which provides the legal framework for the conservation and sustainable use of oceans and their resources, as recalled in paragraph 158 of The Future We Want

Subtarget text taken from sustainabledevelopment.un.org/sdg14.

One stated objective of the conference was to build on existing partnerships and foster new collaborations that focused on ocean issues, including conservation and MPAs (United Nations, 2016). One proposed way to achieve this objective was through the creation of a voluntary commitment program for fulling SDG 14, including all of its 10 subtargets. This "Call for Action" was produced during a February preparatory meeting preceding the Ocean Conference, and came from the heads of state and government, as well as high-level representatives.

Under this Call for Action, developed by nation-state delegations, 22 endeavors were listed for stakeholders to partake in. Among these endeavors was an appeal to create a voluntary commitments database regarding oceans. This database was proposed to be open to anyone, including governments, NGOs, and even individuals. The Call for Action was also published on the official website of the UN Ocean Conference, allowing it to be viewed by those attending the meeting as well as the wider public. During the months preceding the 2017 United Nations Ocean Conference, as well as after, stakeholders were invited to make voluntary commitments under SDG 14. As of September 2017, 1,395 commitments were registered through the voluntary commitment process, spanning across organizations and areas of focus related to SDG 14 as a whole. To date, the voluntary commitment call is still open and accessible, and the website features updates on previously made commitments.

One way to understand where the world is headed in terms of global MPA targets is through dissecting the voluntary commitment process under the 2017 UN Ocean Conference. Here, we analyze a subset of these commitments that are specifically related to SDG 14.5, which focuses on the creation of MPAs. Analyzing the distribution of voluntary commitments surrounding MPAs offers a potential predictor of whether the goal of 10% protection of the oceans will be achieved. Using government commitments under 14.5, we created a map of potential MPA commitments, including those focused on resilient MPAs. An emergent theme from the data was that many of the MPA commitments referenced resilience, but did not define what form of resilience was to be achieved. Resilience can be thought of as a cluster concept, in that it is a word with multiple meanings (Parsons, 1973). Resilience as a benefit of MPAs has been written on extensively, but often lacks an operational definition (Nocito, 2018). Below we present on the overall number of voluntary commitments made, which actors made them, the geographic location of the commitment, and the kind of MPA committed. We then further present on the use of resilience in the voluntary commitments, including which actors focused on resilience and the forms of resilience referred to. Finally, we reflect on the potential strengths and weaknesses of these voluntary commitments in moving forward toward a global system of MPAs.

MATERIALS AND METHODS

To study the voluntary commitment process of the UN Ocean Conference, we completed an empirical textual analysis of the content of the voluntary commitments. To create a voluntary commitment, a member of an organization must fill out a commitment registration form online. Some of the information is open-ended, such as project timeline, partners, and description. Other aspects are preset, such as what aspects of SDG 14 does the commitment concern and what features of an MPA are being committed to. The preset feature, however, also prevented capture of some finer details, such as size of MPA being proposed.

To carry out our textual analysis, we downloaded voluntary commitments[2] related to SDG 14 in September 2017, 3 months after the close of the UN Ocean Conference. This database is publicly accessible. To identify how priorities were distributed over the entirety of SDG 14, we sorted all the commitments by what sub-goals of SDG were selected as being achieved through the commitment. To gain a better understanding of what entities were creating commitments of SDG 14 overall, we then sorted commitments by the nature of the actor making the commitment. Actors include: Government, UN entities, IGOs, NGOs, civil society organizations (CSOs), Academic Institutions, Scientific Communities, Private Sector, Philanthropic Organizations, Partnerships, and Others. This pre-sorting gave us a set of voluntary commitments that were seeking to help implement SDG 14.5.

These 14.5-related commitments were then sorted into those that referred to resilience within the description of the commitment text. Because "resilience" was not a categorized keyword, we searched each individual voluntary commitment text for references to resilience. The various definitions of resilience were developed through a meta-analysis of 183 papers that referenced both resilience and MPAs. Papers were downloaded from Web of Science, a database, using a nested search approach. Nest one included the terms: marine reserve, marine nature reserve, MPA, MPA*, no take reserve, MPA. From that initial search, a secondary nest was created, using the terms: resilien*. This allowed the papers from nest one to be searched for references to resilience, resiliency, and resilient. From the papers, definitions of the various types of resilience were either given or created (Nocito, 2018; **Table 2**). AS validated the codes that were produced. Codes were reviewed three times by the lead author using grounded theory and followed Strauss and Corbin's three step process: open coding, axial coding, and selective coding (Corbin and Strauss, 1990). In cases where papers from the meta-analysis lacked a specific definition within the text, definitions were created by referencing various papers to create a single, salient definition. We then used these definitions to code the resilience sub-set of the voluntary commitments.

To create a map of potential MPAs based on commitments, only national governments were selected, as other groups such as NGOs and CSOs may work in multiple countries and only governments have the authority to designate MPAs. The EU was also omitted for the same reason as it cannot establish MPAs without working through an individual country. Voluntary commitments were re-downloaded in December 2018 to create as recent of a map as possible. Data were sorted by the filters of "14.5" and "government entity."

We quantitatively compared the distribution of the total pool of voluntary commitments among different entities to the distribution of commitments under SDG14.5, in which we assigned expected values for SDG14.5 commitments based on the initial distribution of all voluntary commitments. We also quantitatively compared the distribution of the pool of MPA-related commitments among different entities to the distribution of commitments focused on resilience. For this comparison we assigned expected values for resilience commitments based on the distribution of SDG14.5 commitments. All statistical analyses were performed as two-tailed chi-square tests,

[2]https://oceanconference.un.org/commitments/

TABLE 2 | Definitions of different forms of resilience.

Code	Definition	Source
Biological	"Changes in the structure of natural communities following multiple acute disturbances are often related to inter-specific differences in their ability to resist pressures and/or their capacity to recover in the aftermath of disturbances"	Shedrawi et al., 2017
Biological-Fish	"The ability of an ecosystem that supports large-scale fisheries to adapt, resist, or recover"	Nocito, 2018
Climate	"The ability of an area to either (a) adapt, (b) resist, and/or (c) recover from the effects of climate change or climate variability"	Nocito, 2018
Coastal	"Ability of a community to 'bounce back' after hazardous events such as hurricanes, coastal storms, and flooding"	NOAA, 2017
Community	"The existence, development, and engagement of community resources by community members to thrive in an environment characterized by change, uncertainty, unpredictability, and surprise"	Magis, 2010
Coral	"Refers to the capacity of an ecosystem to tolerate disturbance without abruptly shifting to an alternate regime and losing structure, function, or services"	Abelson et al., 2016
Economic	"A business' ability to adapt and respond to an economic impact"	Moore et al., 2016
Ecosystem	"Measure of the persistence of systems and of their ability to absorb change and disturbance and still maintain the same relationships between populations or state variables"	Holling, 1973
General	"The capacity of a system to continually change and adapt and yet remain within critical thresholds"	Glaser et al., 2015
Other	If no commitment fit into the categories, it was given the code of other	Nocito, 2018
Social–Ecological	"The capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity, and feedbacks"	Walker et al., 2004

These definitions informed the coding process of voluntary commitments referencing resilience under sustainable development goal (SDG) 14.5.

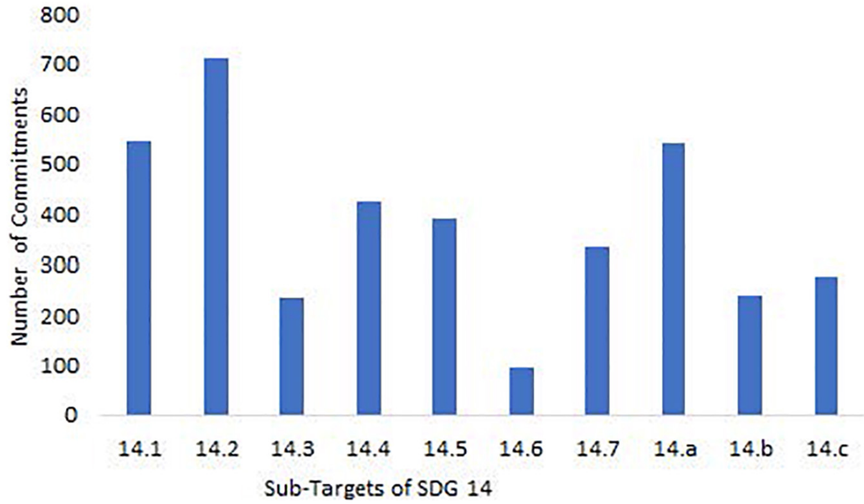

FIGURE 1 | Voluntary ocean commitments. Number of voluntary commitments (*N* = 3,795) made with regards to sustainable development goal (SDG) 14 (Life under water) subgoal. Note that an individual commitment could address multiple specific goals. Data collected from https://oceanconference.un.org/commitments/ between June 2017 and September 2017.

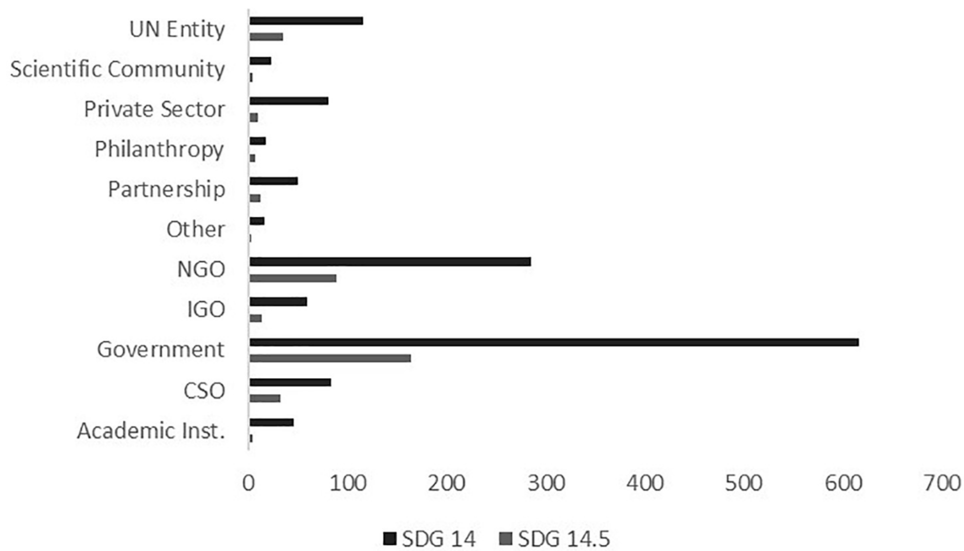

FIGURE 2 | Number of commitments made under SDG 14 and SDG 14.5. Sustainable development goal (SDG) 14 commitments (*N* = 1,395) and SDG 14.5 commitments (*N* = 376) sorted by entity who made the commitment. Data collected from https://oceanconference.un.org/commitments/ between June 2017 and September 2017.

comparing observed distributions using the R statistical software program (base package).

RESULTS

How Many Commitments?

Of the 3,795 subtarget commitments made as of September 2017, only 10% (389 subtarget commitments) commitments pertained to SDG 14.5 (**Figure 1**). SDG 14.2, which pertains to ecosystem-based management of the coastal and marine environment, had the largest portion of the commitments, at 19% (713 subtarget commitments). SDG 14.3 pertains to ocean acidification and accounted for 6% (236 subtarget commitments). SDG 14.4 aims to end illegal, unregulated, and unreported fishing and accounted for 11% (423 subtarget commitments) of the total commitments. SDG 14.6 aims to decrease the number of fisheries subsidies and accounted for 2% (95 subtarget commitments) of the subtarget commitments. SDG 14.7 pertains to increasing the economic benefits for Small Island Developing States and accounted for 9% (335 subtarget commitments) of the total commitments. SDG 14.a aims to increase scientific knowledge

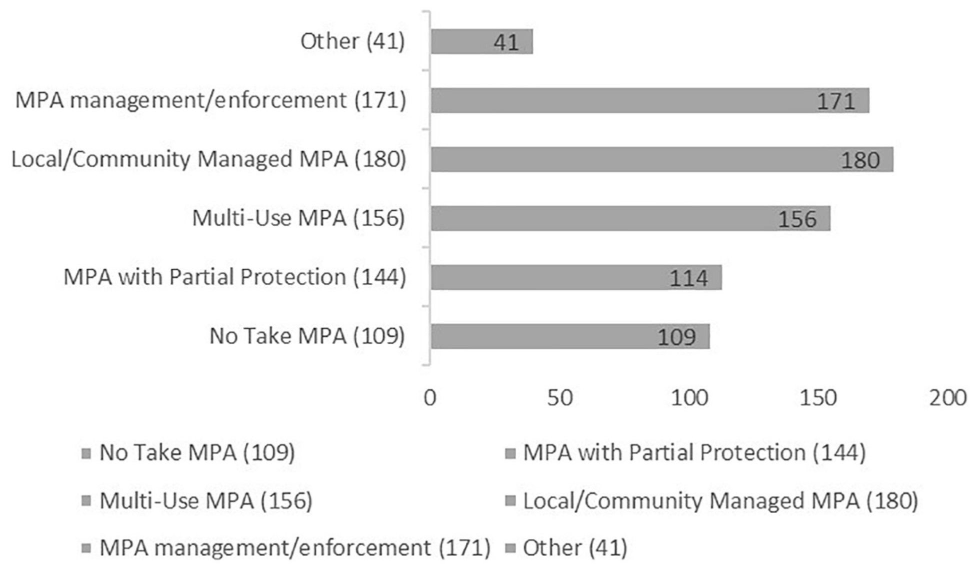

FIGURE 3 | Marine protected area commitments. Voluntary commitments made with regards to sustainable development goal (SDG) 14.5 (marine protected area goal). Chart represents proportion chosen within the different category commitments. Data from https://oceanconference.un.org/commitments/ as of September 2017.

of the marine environment and to develop marine technology and accounted for 14% (541 subtarget commitments). SDG 14.b pertains to small-scale artisanal fishers and their rights to access the marine environment and accounted for 6% (241 subtarget commitments). SDG 14.c encourages governments to implement national laws in line with the UN Convention on the Law of the Sea and accounted for 7% (278 subtarget commitments).

Who Made the Commitments?

Eleven different types of actors or entities made voluntary commitments to SDG 14, including SDG 14.5. A breakdown of which entities made commitments under SDG 14, and SDG14.5 is shown in **Figure 2**. Overall, the distribution of the 376 SDG 14.5 implementing commitments across the 11 entities significantly differed from the distribution of the 1395 voluntary commitments across entities (d.f. = 10, p = 0.04). This difference was driven markedly by the under performance of the private sector and academia and the over representation of NGOs and CSOs.

What Kinds of MPAs Were Proposed in Voluntary Commitments?

The voluntary commitment portal allowed participants to select preset types of MPA commitments (**Figure 3**). Twenty-four percent (180 commitments) of the commitments pertain to local and/or community managed MPAs. Multi-use MPAs accounted for 20% (156 commitments) of the commitments. Fifteen percent (144 commitments) of the commitments concern MPAs with partial protection, which can mean the MPA has features such as seasonal closures or fisheries permits. Only 14% (109 commitments) of the commitments were for no-take MPA. Twenty-two percent (171 commitments) of the MPA commitments are toward supporting management and enforcement of MPAs. The category of "Other" allows the entity to put in any deliverable that is not covered by predetermined categories. Other accounted for the lowest percentage, at only 5% (41 commitments) of the total SDG 14.5 commitments.

What Kinds of Resilience Are Included?

Resilience was coded 132 times over 91 voluntary commitments for SDG 14.5. Climate resilience accounted for one-third (43 mentions) of the total references of resilience (**Table 3**), followed by ecosystem resilience at 17% (22 mentions) (**Figure 4**). Community resilience was accounted for 11% (15 mentions) of the overall references. SES resilience accounted for 8% (11 mentions). Biological resilience accounted for 7.5% (10 mentions), while biological-fish resilience accounts for 5% (7 mentions). General resiliency also made up 5% (7 mentions) of the overall references. Coral resilience accounted for 7% (nine mentions) of the references. Economic resilience accounted for 4% (five times). Coastal resilience only accounted for 2% (three mentions) of the references, although SDG 14 and SDG 14.5 deals with both marine and coastal environments. Lastly, the category of "other" only accounted for >1% (1 mention) of the references.

What Actors Use Which Forms of Resilience in MPA Proposals?

Resilience MPA commitments were made by all of the 11 entities that made commitments under SDG 14.5. NGOs made 22% (20 commitments) of the resilience commitments (**Figure 5**). Consistent with overall trends of entity commitments (**Figure 1**), government is leading the number of 14.5 commitments that reference resilience at 36% (33 commitments). UN entities accounted for 10% (nine commitments). IGOs accounted for

TABLE 3 | Examples of resilience commitments.

Code	Voluntary commitment example	Organization making the commitment
Biological	"Maximize the resilience of vulnerable species to the impacts of climate change and climate variability by reducing other pressures, including poor water quality."	Government of Australia
Biological-Fish	"Promote measures to improve management and resiliency of fisheries/marine resources."	Government of Belize
Climate	"California's evaluation of its MPA Network will include a focus on helping better understand how areas that reduce or remove fishing impacts may respond differently to, and potentially build resilience against, additional stressors like climate change and invasive species."	Ocean Protection Council on behalf of the State of California (Government)
Coastal	"Reduction of land-based marine littering, strengthening the resilience of coastal zones against the impacts of climate change. . ."	Government of Germany
Community	"Monaco commits financially support this integrated approach in favor of ocean acidification monitoring, strategies to strengthen the resilience of local communities, and concrete actions to adapt to and mitigate ocean acidification."	Government of Monaco
Coral	"This will protect coral reef biodiversity; build climate resilience of reefs as well as dependent industries and communities; and make coral reefs a part of sustainable development/a blue economy."	Global Coral Reef Partnership (NGO)
Economic	"Additionally, education and climate financing must also be made available to help developing countries build resilience."	Perfect Union (NGO)
Ecosystem	"Pacific Island communities and ecosystems are resilient to the impacts of ocean acidification and a changing ocean, with practical adaptation measures and alternate livelihoods in place."	Secretariat of the Pacific Regional Environment Programme (IGO)
General	"This initiative aims at conserving and sustainably use our marine environment and its resources for our current and future generations. It is also our contribution to the regional and global effort to maintain and restore the health, productivity, and resilience of our Ocean."	French Polynesia Government
Other	"Art Installations underwater provide opportunities for studies on corals, their evolution, resilience, and species interaction."	Raisa Mar-Conservation Artist (Other)
Social–Ecological	"Build socio-ecological resilience to coral reef degradation in the islands of the Western Indian Ocean."	Plymouth Marine Laboratory (NGO)

Examples of voluntary commitments made under SDG goal 14.5 which represented the various resilience categories (based on ***Table 2****), including which entity made them.*

t 9% (nine commitments). The scientific community, private sector, philanthropy, partnership, and CSOs each accounted for 3% (three commitments each). The entity of "Other" made 2% (two commitments) under SDG 14.5 that referenced resilience.

Different entities focused on different types of resilience in their voluntary commitments. Governments made the most references to resiliency overall, accounting for 37% (49) of the overall references. Government's main focus was on climate resilience over the other forms (**Figure 6**), accounting for 39% (19 references). NGOs accounted for 20% (27 references) of the overall references. Of these, climate resilience accounted for 22% (6 references) of the NGOs total references to resilience. 100% (three references) of the references of coastal resilience were made by governments.

Looking solely at the governments making resilience MPA commitments (**Figure 6**), 39% (19 commitments) referred to climate resilience, followed by ecosystem resilience at 21% (10 commitments). Biological–Fisheries resilience accounted for 10% (5), and community resilience accounted for 10% (five commitments). Coral resilience and coastal resilience each accounted for 6% (three commitments) of the government commitments. Only 2% (one commitment) of the commitments were focused on biological resilience, as well as only 2% (one commitment) referred social–ecological system (SES) resilience. No governments made commitments surrounding the economic resilience of MPAs (0 commitments).

Climate resilience dominated the MPA resilience categories (**Figure 7**). Environmental, which encompasses ecosystem, coral, coastal, biological–fish, and biological forms of resilience accounted for 38% (51 references). Climate, as a single form of resilience, accounts for 32% (43 references) of the references. Social forms of resilience, which include community, economic, and SES, accounted for 23% (31 references). General resilience was singularly grouped, and it only accounted for 5% (seven references), and "other" was singularly coded accounting for $>1\%$ (one reference).

What Actors Use Which Forms of Resilience in MPA Proposals?

Overall, there is a significant difference between the distribution of entities making MPA-related commitments under the voluntary commitment process, and those making specific reference to resilience in their commitments (d.f. = 10, $p < 0.01$). Just as for total MPA commitments, state governments made the greatest number of commitments that incorporated resilience but were actually underrepresented in their use of resilience ($n = 49$, expected = 57). Similarly, NGOs comprised the second largest number of resilience references, but also underperformed ($n = 27$, expected = 31). Academic entities were the greatest over performers when it came to resilience references ($n = 7$, expected = 2). UN entities and IGOs also overperformed in their use of resilience ($n = 14$, expected = 12 and $n = 11$, expected = 4, respectively).

Climate resilience was the most dominant form of resilience across all entities. Thirty-nine percent of the resilient MPA commitments made by governments were related to climate

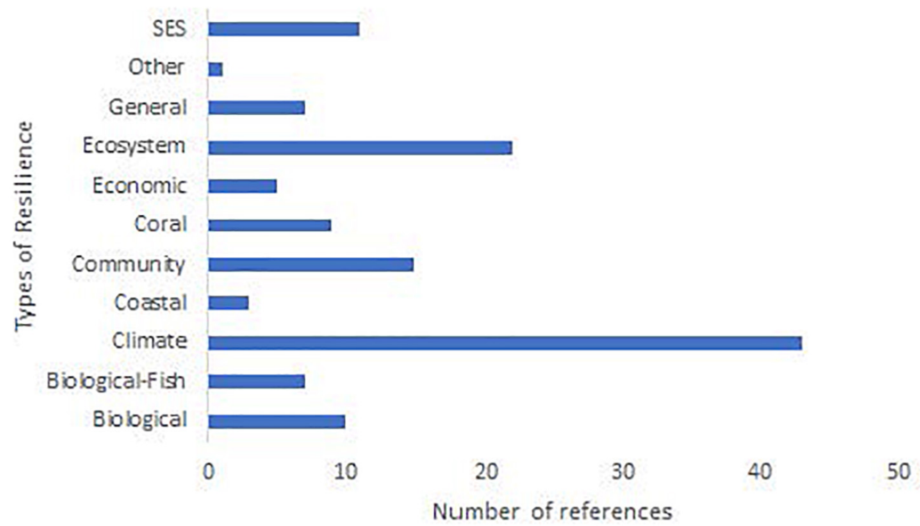

FIGURE 4 | Marine protected area commitments related to resilience. Voluntary commitments made with regards to sustainable development goal 14.5 (related to marine protected areas) which specifically mentions resilience in the descriptor (see **Table 2** for definitions of resilience; **Table 3** for examples of commitments). Data from https://oceanconference.un.org/commitments/ as of September 2017.

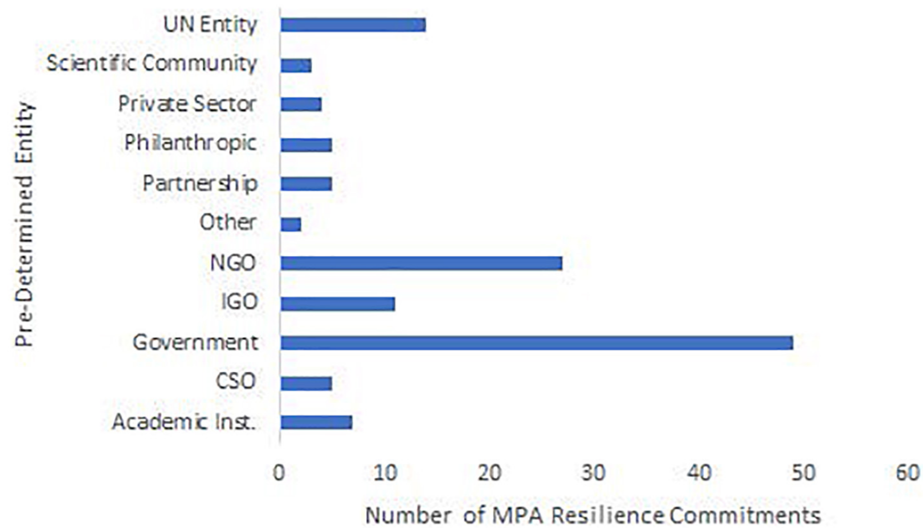

FIGURE 5 | Resilience MPA commitments according to entity. Each commitment description under SDG 14.5 was read for references toward resilience. Data were collected in September 2017. *N* = 91.

resiliency (**Figure 6**). However, there were no commitments made by governments that related to economic resiliency when discussing SDG 14.5, while NGOs did not focus their use of MPAs on coastal resilience. The scientific community was dominated by a focus on biological and climate resilience (**Figure 6**).

Where Are the MPA Commitments Being Made?

Multiple nation states made voluntary commitments toward MPAs under SDG 14.5 (**Figure 8** and **Supplementary Table S1**). Sixty-five nation-states governments committed to created MPAs, for a total of 166 potential MPAs. Sweden led the way with 10 voluntary commitments (6%) toward creating MPAs, followed by Canada with 8 voluntary commitments (5%). Pacific Small Island Developing State (PSIDS), as a whole, made 34 voluntary commitments (20%) toward creating MPAs.

DISCUSSION

The UN Ocean Conference brought the marine environment to the forefront of international issues. For the first time, various sectors came together to discuss issues surrounding the oceans,

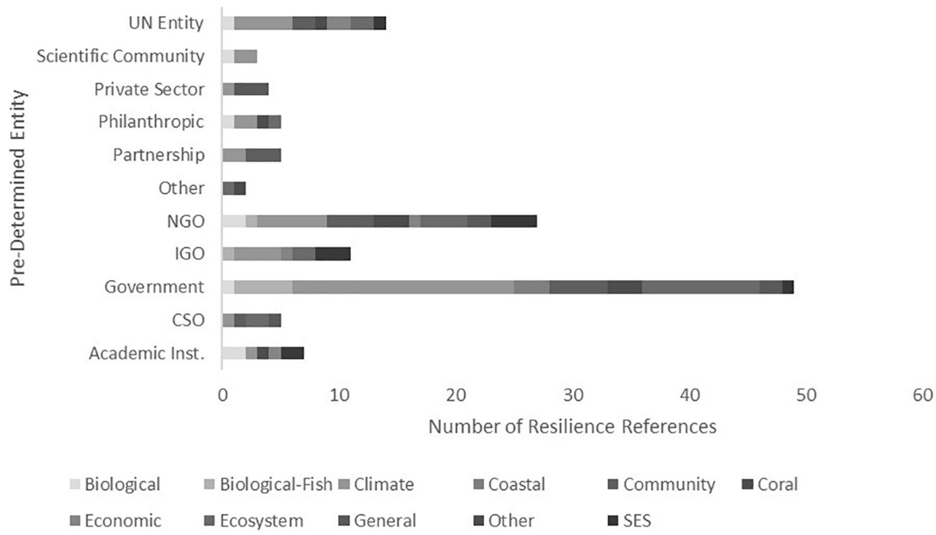

FIGURE 6 | Resilience codes by entity. The panel shows both the number of references made by each entity and the breakdown of what type of resiliency each entity made. See **Table 3** for resiliency definitions. $N = 132$. Data collected from https://oceanconference.un.org/commitments/ as of September 2017.

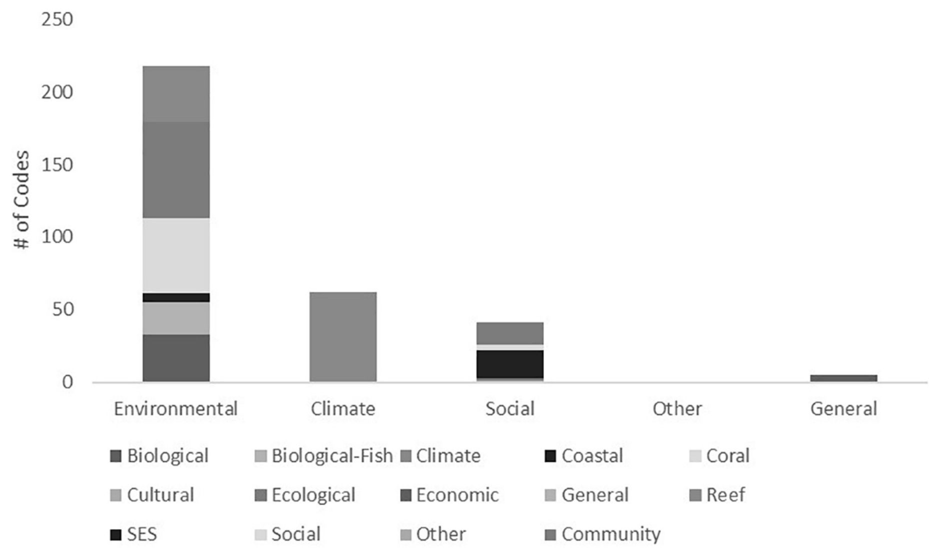

FIGURE 7 | Grouped types of resilience from all entities made in voluntary commitments of SDG 14.5. $N = 133$. Climate is single coded to emphasize its number and overall importance to entities. Data collected from https://oceanconference.un.org/commitments/ as of September 2017.

including its conservation and sustainable use of the marine and coastal environment. This is key given international targets and imminent timelines: to conserve 10% of the marine environment by 2020 and to establish a new treaty guiding MPA designation processes on the high seas by 2020. The large number of voluntary commitments made aligns with these global trends (Lubchenco and Grorud-Colvert, 2015; Boonzaier and Pauly, 2016).

Despite much interest by non-state actors (e.g., NGOs, foundations, and the private sector) in taking the mantle of conservation leadership, national governments still made the largest number of commitments. This is consistent with governments being the only entities with the authority to establish and implement MPAs (Agardy, 1994). Thus, the responsibility to fulfill global commitments rests on them. Yet NGOs also made their fair share, showing the commitment of external organizations in working with governments and communities toward developing a global network of MPAs (Christie and White, 2007; White et al., 2010). In our MPA crystal ball, there is a clear indication of the large role of non-state actors in the development of future MPAs, but no indication that establishment and implementation of MPAs will become anything other than a state-led process.

The voluntary commitments toward MPAs ranged from no-take commitments to multi-use and community MPAs. This

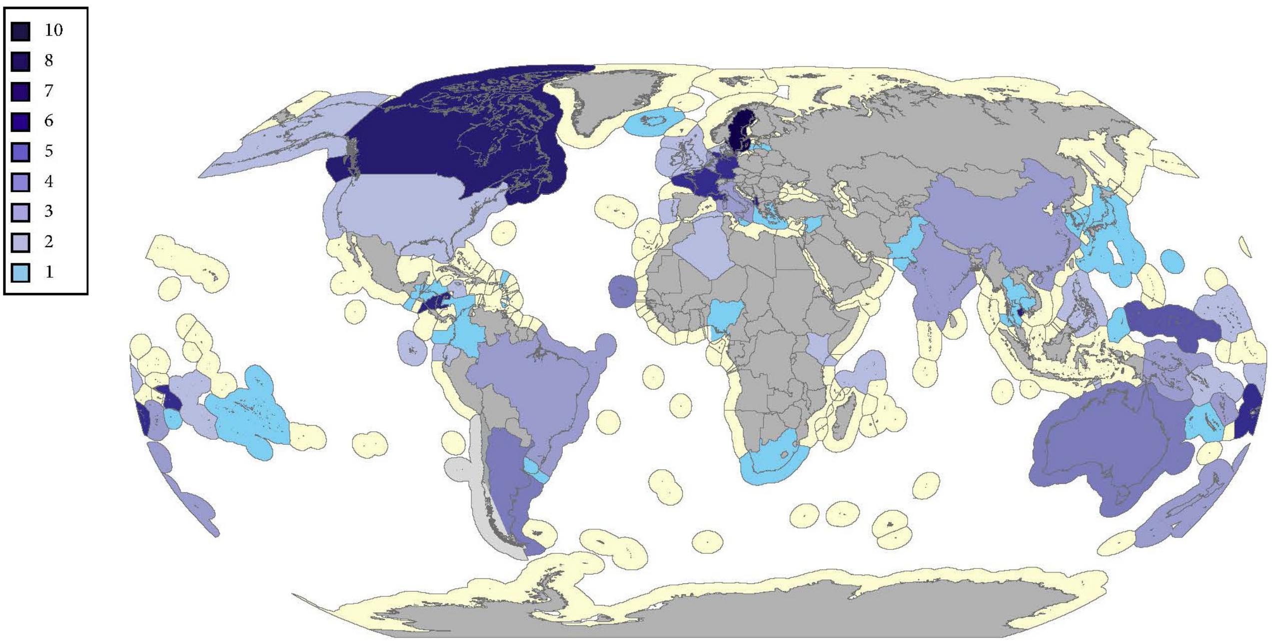

FIGURE 8 | Map of government's MPA voluntary commitments (SDG 14.5) from 2017 UN Ocean Conference. This map shows proposed areas of new MPAs, within the country's exclusive economic zone. *N* = 65 countries, not including the EU. 173 potential MPA sites were identified. Data collected from https://oceanconference.un.org/commitments/ as of November 2018.

reflects the complexity inherent in MPAs where trade-offs must be made between conservation and allowing for sustainable use (Hirsch et al., 2010; McShane et al., 2011; Davies et al., 2018). This may also reflect the multiple stakeholders involved in developing MPAs at the national level (Brown et al., 2001). Importantly, having stakeholders, such as the fishing industry, involved in citing MPAs can lead to higher levels of compliance (Oracion et al., 2005) yet it may lead to a less ecologically effective MPA. No-take MPAs, where no use is allowed, have been shown to be the most effective at conserving biodiversity (Lester et al., 2009; Edgar et al., 2014; Costello and Ballantine, 2015; Sala and Giakoumi, 2018). Also, some of the MPA categories, such as multi-use or community, may not even qualify as an MPA. Under internationally recognized IUCN guidelines, community-managed protected areas that are managed mainly for the extraction of marine genetic resources should not be automatically classified as an MPA (Dudley, 2008; Day et al., 2012). In our MPA crystal ball, there is a clear indication that MPAs are no longer just about fisheries conservation.

This research also showed that many entities view the creation of MPAs as a path toward resilience. One of the most highlighted goals of MPAs in recent studies are their role in enhancing resilience (Barnett and Baskett, 2015; Hopkins et al., 2016; Mellin et al., 2016). In particular, the voluntary commitment process involved a heavy focus on MPAs as tools of climate resilience, aligning marine conservation with broader discussions about trajectories for global climate policy. Climate resilience accounted for one-third of the total mentions of resilience, followed by ecosystem resilience at 17% of the mentions (**Figure 4**). Also, academic entities made the greatest number of references to resilience, which is perhaps to be expected, given the academic origins of the resilience concept. All of the references of coastal resilience were made by governments, which is in line with government priorities of their exclusive economic zones, which are located within 200 nm of a nation-states coastline.

Given that the bulk of the literature deals with ecosystem resilience, there may be a paradigm shift toward climate resilience occurring in terms of practical applications of resilience. This is in line with increasing evidence that MPAs can enhance resilience of marine systems under environmental change and stress (Olds et al., 2014; Mellin et al., 2016; Roberts et al., 2017; Darling and Côté, 2018; Laffoley et al., 2019). The increased focus on resilience in MPAs shows where priorities may lie, such as on resilient fisheries or resilience toward climate change (McClanahan et al., 2012; McLeod et al., 2012; Green et al., 2014; Ford et al., 2016). In our MPA crystal ball, the future aligns marine conservation, ocean conservation, and climate change global priorities, using the lens of climate resilience as a key organizing principle.

Resilience is understood as a key organizing and framing concept that shapes a systems ability to respond to external stresses, and it is a concept widely deployed in adaptation science, ecological science, and common pool resource management theory (e.g., Holling, 1973; Tompkins and Adger, 2004; Mosimane et al., 2012). But what kinds of resilience are being discussed in voluntary commitments? This research revealed the wide array of interpretations of resilience across entities, in line with the lack of clarity around this term in the literature. Nocito (2018) found that the amount of literature surrounding MPAs and resilience has steadily increased since the 1990s, but that only one-third of the papers gave a definition of what form of resilience the authors were referring to within the text. This is concerning when resilience is considered an aim or a goal of a MPA, as without proper definition the success of reaching that aim may come into question.

The voluntary commitments also provided a way to map out potential future MPAs within country's exclusive economic zones. This mapping exercise shows what countries are – and possibly more importantly are not – pledging future MPAs. While each commitment has its own fulfillment date, this map will help predict where MPAs in exclusive economic zones will exist in the future. The majority of commitments were made in the Pacific, which is expected as the UN Ocean Conference was influenced greatly by PSIDS, as well as co-hosted by the Pacific country of Fiji. The single country making the most MPA commitments was Sweden, totaling in at 10 MPA commitments, followed by Canada at 8 MPA commitments. However, as a region, PSIDS proposed 32 MPAs within the voluntary commitment system. The government of Sweden has committed to fulfilling SDG 14 and Aichi Target 11 through their national legislation body, called the Riksdag (Government of Sweden, 2015). To date, Sweden has 1,373 MPAs in their waters, making them a leader on MPAs in the EU as a whole (see text footnote 1) (European Environment Agency, 2015). The PSIDS, as a unit, are harbor 466 MPAs to date (see text footnote 1). PSIDS have called for a strong commitment to MPAs in international dialogs, and emphasize their commitments previously to creating MPAs (Moses, 2017). In our MPA crystal ball, based on the voluntary commitment process, MPAs are increasingly a tool of wealthy, conservation minded developed countries, and small island states. The lack of commitment from major emerging economies is a sign that work is to be done to build a broader coalition of economic and political leaders for conservation (Miller, 2014).

Ultimately, despite slow progress on achieving global MPA goals, it is clear that the use of area-based management tools as policy instruments to provide protection for oceanic spaces is an idea that is not going away (Boonzaier and Pauly, 2016). Yet the idea of MPAs, like all policy ideas that have come into maturity through implementation, is evolving. It is moving toward the incorporation of a multi-sector, multi-stakeholder approach in MPA development and in the proposal process for MPAs. MPAs are now fully understood to be tools of climate change resilience, yet, ultimately, their success must still be measured by the efficacy of their implementation for achieving an increasingly broad set of policy-goals.

Sustainable development goal 14 is set to expire in 2020. By that time, the goal is to have 10% of the marine and coastal environment conserved through area-based management tools, such as MPAs. To date, only 4.8% of the global ocean is conserved (see text footnote 1). While these predicted MPAs will add to that, there is still a lot of work to be done to reach 10%. These countries need to act quickly to create and establish these proposed MPAs by 2020, for the goal to be met. Countries must also consider that not all MPAs are created equal. No-take MPAs, or marine reserves, are often seen as the strongest MPAs for conservation and restoration of ocean processes (Russ and Alcala, 2004; Lubchenco and Grorud-Colvert, 2015). If the majority of these proposed MPAs are multi-use, or with partial protections, it will still go toward that 10% goal. However, the benefits derived from them may be less than expected since they are still being used and subject to anthropogenic stress (Lester and Halpern, 2008).

CONCLUSION

With various international goals and targets aimed at reaching 10% of the marine environment conserved through MPAs by the year 2020, it is fitting that so many entities have turned their attention toward fulfillment. The voluntary commitment portal of the UN Ocean Conference allows these entities to receive well-earned attention of their efforts. From these voluntary commitments came a newfound movement toward resiliency, but also showed the dire need of operational definitions to ensure success. The different types of resilience show what types are being prioritized, and by whom. The voluntary commitment portal also allowed a map of potential future MPAs to be created. This map shows which countries are truly committed to fulfilling SDG 14.5, and emphasizes how few countries actually made SDG 14.5 commitments through the voluntary commitment portal.

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the article/**Supplementary Material**.

AUTHOR CONTRIBUTIONS

EN, AS, and CB designed the research and wrote the manuscript. EN and AS carried out the research and conducted the analyses.

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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.2019.00835/full#supplementary-material

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Marine Conservation Begins at Home: How a Local Community and Protection of a Small Bay Sent Waves of Change Around the UK and Beyond

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The Firth of Clyde, on the west coast of Scotland, was once one of the most productive fishing grounds in Europe. However, successive decades of poor management and overfishing led to a dramatic loss of biodiversity and the collapse of finfish fisheries. In response, concerned local residents on the Isle of Arran, which lies in the middle of the Clyde, formed the Community of Arran Seabed Trust (COAST) in 1995. After 13 years of campaigning, a small (2.67 km^2) area in Lamlash Bay became Scotland's first no-take zone (NTZ) in 2008, and only the second in the UK. Since protection, biodiversity has increased substantially, along with the size, age and density of commercially important species such as the king scallop, *Pecten maximus*, and the European lobster, *Homarus gammarus*. Arguably more important, however, is the influence the Lamlash Bay NTZ and COAST have had on UK marine protection in general. Most notably, detailed research has created a case study that clearly demonstrates the benefits of protection in an area where little such evidence is available. This case has been used repeatedly to support efforts for increased protection of UK waters to help rebuild marine ecosystems and enhance their resilience in an uncertain future. In Scotland specifically, lobbying by COAST led to the designation of a much larger marine protected area (MPA, $>$250 km^2) around the south of Arran, one of 30 new MPAs in the country. Evidence from Lamlash Bay has supported development of strong protection for these MPAs, seeing off lobbyist efforts to weaken management. Arran's conservation success has been recognized internationally and is inspiring greater involvement of local communities around the UK, and further afield, to take the destiny of their coastal waters into their own hands. Successful marine conservation begins at home.

Keywords: marine protected areas, marine reserve, community based conservation, ecosystem – based management, fisheries, marine biodiversity, Lamlash Bay, Isle of Arran

INTRODUCTION

Despite a recent increase in the coverage of Marine Protected Areas (MPAs), improvements in fisheries management (Worm et al., 2009), and ambitious international agreements for conservation (CBD, 2010; U.N, 2015), marine biodiversity continues to decline worldwide (WWF, 2018). The global degradation of marine ecosystems can reduce the number and quality of ecosystem goods and services they provide, negatively affecting human livelihoods and well-being (Naeem et al., 2016). Reductions in biodiversity can also leave marine ecosystems less resilient to future shocks and changes (Howarth et al., 2014). Furthermore, rapidly increasing threats from ocean warming and acidification could degrade marine ecosystems further unless there is rapid action to reduce greenhouse gas emissions (Nagelkerken and Connell, 2015).

A key reason why conservation targets have failed to protect biodiversity is that they can promote responses that focus on giving the appearance that action has been taken, rather than ensuring the action taken was effective. For example, a major driver of the recent increase in MPAs was the CBD Target 11, set in Nagoya in 2010, to ensure that '*at least 10% of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas*' (CBD, 2010). In response, many countries rapidly designated new MPAs within their waters, without necessarily using the best available science to design them, or giving sufficient prior thought as to how they will be managed or enforced (Lubchenco and Grorud-Colvert, 2015). Many countries have also attempted to meet conservation targets by designating MPAs in remote overseas territories, as these usually receive less opposition than MPAs proposed around the mainland (Jones and De Santo, 2016).

Although some MPAs in overseas territories have considerable conservation value (O'Leary et al., 2018), these designations may occur at the expense of providing MPAs closer to home, in places where they are greatly required because of intense human activity (Jones and De Santo, 2016). For example, the UK has now protected 1.5 million km^2 of ocean in UK Overseas Territories (UKOTs) where fishing is completely banned, with a further 1.5 million km^2 under partial protection and more promised in the future (O'Leary et al., 2019). In contrast, there are only four highly protected MPAs around the British Island, out of a total of 355, which cover just 21.07 km^2 or 0.0024% of UK seas (July 2019: Howarth et al., 2011; Solandt, 2018; JNCC, 2019). Fewer than half of these MPAs have management plans (JNCC, 2019), and most these are unambitious, with aims to maintain degraded ecosystems in their present state (Plumeridge and Roberts, 2017). Consequently, most UK MPAs still allow fishing within their boundaries, including damaging methods such as scallop dredging and bottom trawling (JNCC, 2019).

This problem of 'paper parks,' which give the appearance of protection without actually delivering it, is a global one. In fact, a recent study of MPAs in European seas by Dureuil et al. (2018) found trawling activity and declines of elasmobranchs to be higher inside MPAs than outside, because management was either inappropriate or poorly enforced. 'Paper parks' are problematic because they allow authorities to claim they have taken effective action when the reality is untrue. MPAs may become paper parks if (a) they are poorly sited and/or designed, (b) management plans fail to address present and future threats to biodiversity, and (c) are poorly enforced (Rife et al., 2013). From the opposite perspective, a global review by Edgar et al. (2014) concluded that successful MPAs are: (1) no take, (2) well-enforced, (3) old (>10 years), (4) large (>100 km^2), and (5) isolated by deep water or sand.

A recurring theme in the failure of MPAs is lack of stakeholder engagement (Giakoumi et al., 2018). Stakeholders commonly includes fishers, divers and other water users, but may also include concerned local residents and/or visitors. Such people can help provide valuable information about biodiversity at a site and the type and level of human activity present; information needed for effective MPA design (Gleason et al., 2010). Involvement of stakeholders also fosters a sense of stewardship in people, which helps with compliance of regulations and reduces enforcement costs (Giakoumi et al., 2018).

The involvement of stakeholders, including local communities, in MPA designation has been most common in tropical less developed countries (Jones, 2014). In these areas, MPA practitioners have often aimed to work within traditional frameworks of marine resource use and protection (Jupiter et al., 2014). In contrast, the designation of MPAs in developed countries generally follows a top–down process, initiated and controlled by government agencies or other relevant authorities (Jones, 2012, 2014). Stakeholders may be consulted as part of this process, but their overall influence on decisions is often marginal (Jones, 2012). However, in the case study presented here, we describe an exception to this trend whereby the community on Scotland's Isle of Arran changed the course of marine conservation in their local waters, and in doing so, influenced national policy and action. Here, a community identified that their local marine environment was not what it once was, diagnosed the cause and began to campaign for a solution. Arran represents an inspiring case of a tenacious community that would not give up on their local environment. The community on Arran also engaged with scientists early in the process. Science informed their campaigning, and campaign goals helped refine science needs and secure funding. The relationship has been symbiotic and is an excellent example of how science and communities can support each other to achieve the mutual goal of influencing policy to achieve better management. We hope this story will help encourage other coastal communities to take more control over the destiny of their local marine environments to turn both local and global marine conservation efforts toward a more positive direction.

THE FIRTH OF CLYDE – A TRANSFORMED ECOSYSTEM

The Isle of Arran sits in the middle of the Firth of Clyde, off the west coast of Scotland (**Figure 1**). This fjordic

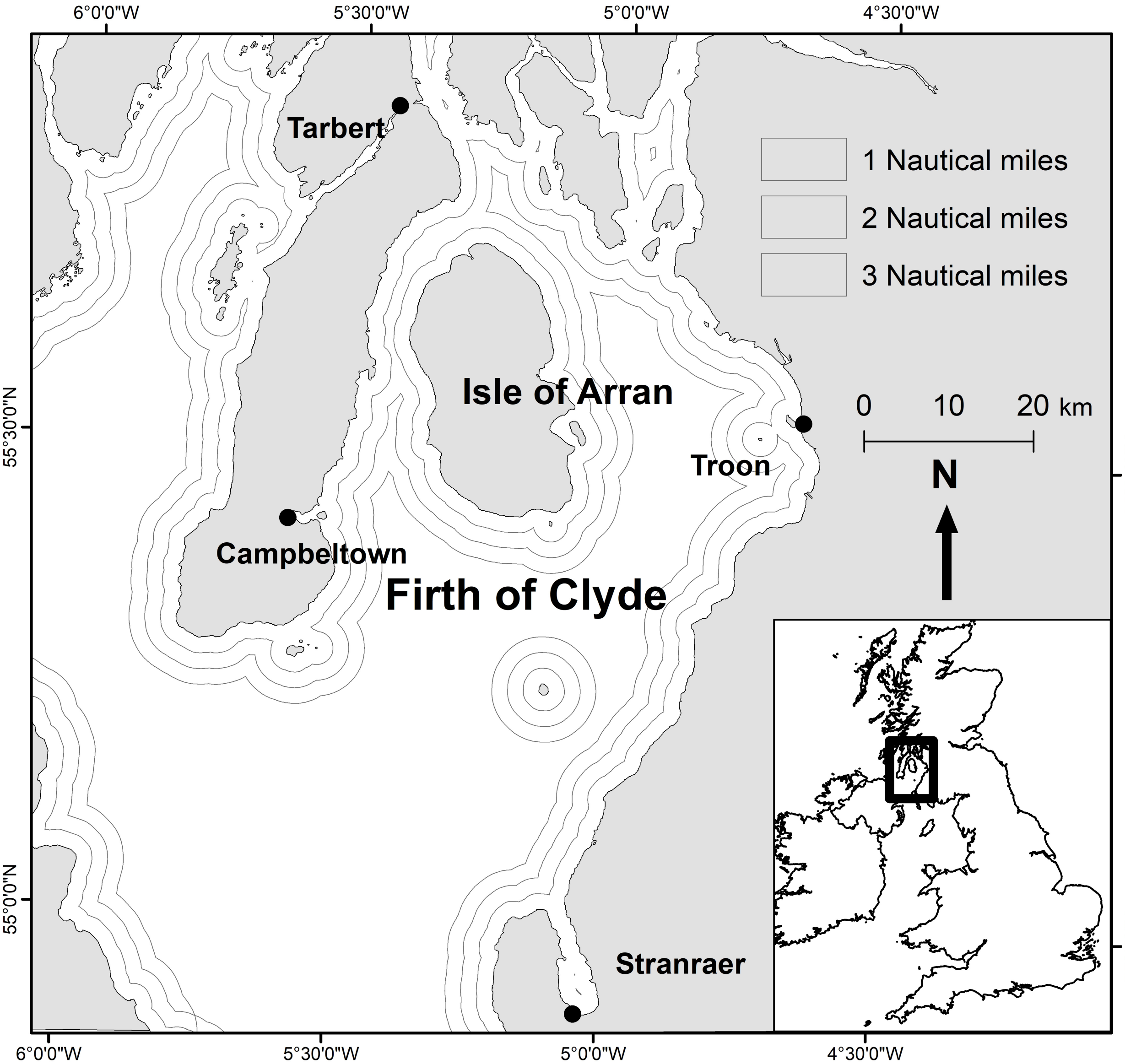

FIGURE 1 | Location of the Isle of Arran in the middle of the Firth of Clyde, Scotland. Inset shows the location of the Clyde within the United Kingdom.

inlet is approximately 100 km long and has supported diverse and highly productive fisheries for species such as herring (*Clupea harengus*), cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*), turbot (*Psetta maxima*), and flounder (*Platichthys flesus*) (Thurstan and Roberts, 2010). However, a series of poor fishery management decisions caused finfish stocks in the Clyde to decline from the mid-1980s. Perhaps the most important of these was the removal of a long-standing ban on trawling by large vessels in the Clyde in 1962, followed by lifting the closure to trawling within 3 miles of shore in 1984 (Thurstan and Roberts, 2010; Heath and Speirs, 2011).

In place of finfish fishing, shellfish fisheries have grown, particularly for Dublin bay prawns (*Nephrops norvegicus),* king scallops (*Pecten maximus)* and to a lesser extent European lobsters (*Homarus gammurus*) and brown crabs (*Cancer pagurus*) (Thurstan and Roberts, 2010). Together these shellfish species now constitute 98% of commercial landings in the Clyde and have an economic value similar to that of the previous finfish fisheries (Howarth et al., 2014). Although Dublin bay prawns and king scallops can be caught using gear that has relatively low environmental impacts (creels and SCUBA diving respectively), in the Clyde prawns are mainly targeted by otter trawls and scallops by Newhaven dredges (McIntyre et al., 2012). These bottom-towed gears can generate considerable bycatch and cause loss of diversity in benthic communities and homogenization of seabed structure (Bergmann et al., 2002; Kaiser et al., 2006; Stewart and Howarth, 2016). Such changes to the Clyde ecosystem, particularly loss of complex habitat essential for survival of juvenile fish, and the lucrative nature of the current shellfish fisheries, are thought to be barriers to the area's restoration (Howarth et al., 2014). Furthermore, this simplified ecosystem and the fisheries it supports are likely to be at risk, both economically and ecologically, from future anthropogenic climate change and the threats it brings (Howarth et al., 2014).

Analysis of fisheries independent surveys demonstrates significant changes to fish community structure in the Clyde between 1927 and 2009 (Heath and Speirs, 2011; McIntyre et al., 2012). During this period, fish biomass changed from being distributed across a large number of species, with many individuals reaching large sizes, to being dominated (90%) by small whiting (*Merlangius merlangus*) below commercial size (Heath and Speirs, 2011; McIntyre et al., 2012). Evidence suggests that high fishing pressure, including bycatch in the current Nephrops fishery, has induced maturation to occur earlier and at smaller sizes in cod, haddock and whiting (Hunter et al., 2015). Baited Underwater Remote Video (BRUV) surveys around the Isle of Arran have also found that whiting recruited earlier, grew faster and were behaviorally dominant over other gadoid species, providing them with a competitive advantage (Elliott et al., 2018). Such evidence at least partly explains the present domination of small whiting in Arran's waters, and suggests that radical management interventions will be needed to restore the Firth of Clyde marine ecosystem to its more diverse and resilient former state.

THE STORY BEHIND THE COMMUNITY OF ARRAN SEABED TRUST (COAST)

The Campaign for Scotland's First No-Take Marine Reserve: 1989–2008

The 20th century transformation of the Clyde was not only recognized by fishermen and scientists, but also by local communities who were frustrated by the apparent lack of intervention by the Scottish government. The Isle of Arran became the focal point for these arguments after local resident and SCUBA diver, Don MacNeish, returned from a trip to New Zealand in 1989. While in New Zealand, Don visited some of the world's first marine reserves and was informed about how they were benefiting fisheries and the wider marine environment (Ballantine, 2014; Whiteside, 2018). Don returned to Arran and suggested to fellow SCUBA diver, Howard Wood that they should try to replicate the same management approach in their local waters as marine reserves were non-existent in the UK at this time.

Together, Howard and Don formed a group known as the Community of Arran Seabed Trust (COAST) in 1995. COAST began to organize meetings with local fishermen and community groups to raise awareness about the deterioration of Arran's marine ecosystem, and to search for solutions. Due to the failure of previous top–down attempts such as the Nature Conservancy Council's 1990 proposal for a Marine Nature Reserve in Loch Sween Argyll in Scotland (Jones, 1999), Howard and Don were determined to build bottom–up, community support for their ideas. Hence, they contacted the then Scottish Office Minister Lord Lindsay to highlight the need for a marine reserve and to show that Government were failing in their duty to protect the natural environment and therefore the public good.

In 1998, during a seminal meeting with local Arran-based commercial fishermen, a community proposal to create a small, fully protected area [No-take Zone (NTZ)] in Lamlash Bay was agreed (Whiteside, 2018). The relevant documentation was presented to the Inshore Fisheries Department (in 1999) in order to:

- Make Lamlash Bay a Marine Protected Area, from which mobile fishing gear is prohibited to regenerate and enhance local fish and shellfish populations
- Establish the first No Take Zone in Scotland, to protect the maerl beds present
- Investigate the fishery benefits of a NTZ and MPA, particularly with regard to scallops.

By protecting and ultimately regenerating maerl, other seabed habitats, fish and scallop populations in Lamlash Bay, COAST's aims were to:

- Improve the local marine environment;
- Help sustain livelihoods of those dependent on fishing and tourism;
- Increase the popularity of Arran as a diving and tourist destination;
- Reverse the dramatic decline of local finfish stocks;
- Educate future generations on the need for marine conservation.

Prior to University and Government involvement, COAST began to document the species and habitats within Lamlash Bay to strengthen the evidence base for protection. Their SCUBA diving members frequently reported the presence of extensive maerl beds (*Phymatolithon calcareum*), a habitat recognized within environmental legislation as being important for biodiversity and fisheries recruitment (Hall-Spencer and Moore, 2000; Kamenos et al., 2004; JNCC, n.d.). COAST therefore used the presence of maerl as the basis of their argument for protection. However, the Scottish Government, and various fishing organizations, claimed their evidence was amateur, biased and unscientific. Hence, with the aim of improving the scientific robustness of their personal observations, eight members of COAST underwent Seasearch training[1] in the spring of 2003. By the end of the summer, COAST had conducted 21 Seasearch surveys within Lamlash Bay (Duncan, 2003). These initial surveys indicated the presence of extensive maerl beds and large abundances of juvenile scallops (Duncan, 2003).

In 2004, Scottish Water announced plans to build a sewage treatment plant adjacent to the proposed NTZ in Lamlash Bay, the effluents of which would be discharged on to the maerl beds. However, thanks to COAST's Seasearch survey reports, Scottish

[1] www.seasearch.org.uk

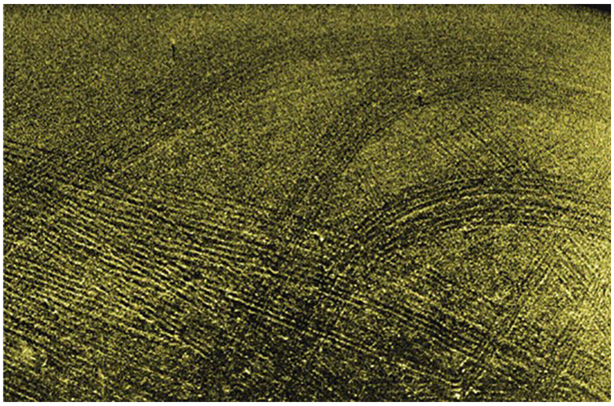

FIGURE 2 | Side-scan sonar images from within Lamlash Bay in 2004, indicating marks created by scallop dredges. Hardly any part of the seabed appears to have escaped impact.

Water were forced to commission an independent survey by The University Marine Biological Station Millport (UMBSM) to confirm the presence of maerl. These new surveys verified COAST's earlier observations and the independent report went on to state: *"In terms of the Clyde Sea, this is highly unusual and damage of that deposit may be damaging one of, if not the last > 90% live Maerl beds in the Clyde sea area. In our expert view, we would strongly advise that this site be avoided by any development which impacts on the sea bed"* (Kamenos et al., 2004). COAST used the report to engage local and national media, raising attention of both the environmental threat to the maerl bed, and to the pursuits of COAST more broadly. In response to this pressure, Scottish Water re-routed the proposed site for the sewage outfall pipe outside of Lamlash Bay. Later in 2004, COAST and UMBSM received funding from the Esmeé Fairbairn Foundation to conduct side-scan sonar surveys of the bathymetry in Lamlash Bay. These surveys revealed large areas of the seabed to be covered in large, sweeping, parallel marks from scallop dredges (**Figure 2**). By combining these striking images with photos of the maerl beds and other habitats within Lamlash Bay, COAST created a powerful visual argument which could easily be communicated to the public, stakeholders and managers, for Lamlash Bay to be designated as an MPA, with an NTZ inside it.

Although much of the research conducted within Lamlash Bay between 2003 and 2006 was spatially and temporally limited, these surveys proved essential to COAST as they provided imagery and data to support their presentations, interviews, campaigns, and proposals. It helped educate the members of COAST and shaped them into a respected, well-known organization. In addition, every survey, no matter how small, attracted more attention and subsequent research.

COAST's campaign began to attract high profile national media attention (see section 'Public Outreach'), with the first significant engagement being featured in a BBC documentary 'Bee in your bonnet.' In 2004, COAST took their concerns to the heart of the Scottish Parliament through submission of a Petition that was supported by many members of the public, marine scientists from all over the world, Members of Scottish Parliament (MSPs) representing all political parties, and many others. The Petitions Committee, after lengthy and careful consideration, passed the Petition to the Environment and Rural Development Committee of the Scottish Parliament where, after a formal enquiry in September 2006, full backing was given for COAST and the Clyde Fishermen's Association (CFA) to develop a proposal for a NTZ in Lamlash Bay. The CFA, who represented most of the mobile (prawn trawling and scallop dredging) fishermen operating in the Clyde, were initially supportive of the NTZ but became increasingly unhappy with the boundaries proposed by COAST. Despite the CFA withdrawing their support for the NTZ in early 2008, a Scottish Government consultation on the NTZ in spring 2008 received 675 responses, 99.3% of which were positive. Finally, after 13 years of campaigning and building community and scientific support, the NTZ in Lamlash Bay was designated on the 20^{th} of September 2008. It was small (only 2.67 km^2 in area) but significant, being the first and still the only fully protected marine reserve in Scotland (second in the UK). Furthermore, it was unique because it was proposed and delivered through local community support, and unlike other UK MPAs, is specifically designed to provide benefits for both fisheries and conservation.

Expanding COAST, the South Arran Marine Protected Area, and Beyond: 2008 – Present

In late 2010, COAST received funding to enable them to appoint their first member of staff, Andrew Binnie, based in a rented office in Lamlash in June 2011. This, along with an assistant appointed in 2012, allowed COAST and its many volunteers to considerably raise its outreach activities and prepare a proposal for a larger MPA around the south of Arran (**Figure 3**).

Concerns raised about the degradation of the Firth of Clyde marine ecosystem by COAST and others (e.g., Thurstan and Roberts, 2010), also began to have wider impacts. In September 2010, COAST, with the help of local MSP Kenneth Gibson, met the Scottish government Cabinet-Secretary for Rural Affairs and the Environment (including fisheries) Richard Lochhead at Holyrood to voice concerns over the decline of the Clyde ecosystem. At this meeting Mr. Lochhead gave a commitment that the Scottish Government would commission their own study into the state of the Firth of Clyde.

The Clyde Ecosystem Review (McIntyre et al., 2012), largely confirmed previous findings (Thurstan and Roberts, 2010). This report was followed by the Clyde 2020 Summit, initiated by Mr. Lochhead in April 2014. This event was opened by the Minister and brought together over 100 stakeholders with interests in the Clyde, to discuss ideas for how to improve marine management. The 'trailblazing' work of COAST and the Lamlash Bay marine reserve was highlighted in his opening address (Marine Scotland, 2014). Following this event, the Clyde Marine Planning Partnership[2] was established in order to take forward

[2]www.clydemarineplan.scot/

the ideas from the Summit. In March 2019, this project released a pre-consultation draft Clyde Regional Marine Plan, designed to significantly improve the health of the Clyde marine ecosystem.

In parallel with the above specific actions in the Clyde, the Scottish government has been working to implement MPAs throughout the country under obligations from the UK Marine and Coastal Access Act 2009 and the Marine (Scotland) Act 2010. Following several years of largely government led research and an extensive consultation, 30 nature conservation MPAs were designated in Scottish waters in August 2014, perhaps the highest profile of which was the South Arran MPA. This multi-use MPA was unique because it was again proposed by COAST, to protect vulnerable seabed habitats, and covers an area just over 250 km^2, including Lamlash Bay (COAST, 2012; **Figure 3**).

Given the uniqueness of the Lamlash Bay NTZ as the only highly protected marine reserve in Scotland, and the availability of scientifically documented evidence of ecological recovery within the NTZ boundaries (**Figures 4**, **5**, Howarth et al., 2011, 2015a,b, 2016), the NTZ played a key role in making the case for designating the South Arran MPA and other MPAs throughout Scotland (COAST, 2012). More significantly, COAST's campaigns and collaborations with scientists have been vital for ensuring the new MPAs receive adequate protection, ensuring a lasting and far-reaching legacy. The Scottish government consulted on the management measures inside 17 of the new MPAs during 2015 (the first of two planned batches, Scottish Government, n.d.). A highly contentious issue was whether to allow trawls and scallop dredges, or only low impact creeling and diving, to continue in these areas. Research from Lamlash Bay, showing recovery of benthic biodiversity in the absence of towed fishing gear, was frequently cited in the consultation responses (Marine Scotland, 2015).

Despite considerable pressure from mobile gear fishers to allow their practice to continue in large areas of the MPAs, the Scottish government banned scallop dredging throughout the South Arran MPA in February 2016. The government agreed to only allow trawling in the outer regions, and implemented similar measures in the other 16 Scottish inshore MPAs and Special Areas of Conservation (SACs) under consideration (Scottish Government, n.d.). In addition, although creeling is allowed in much of the South Arran MPA, it was banned in several particularly sensitive seabed areas (**Figure 2**). This bold protection of the South Arran and several other inshore MPAs by the Scottish government withstood a last-minute challenge from the mobile gear fishing industry in January 2016, which was rejected by Rural Affairs and Climate Change

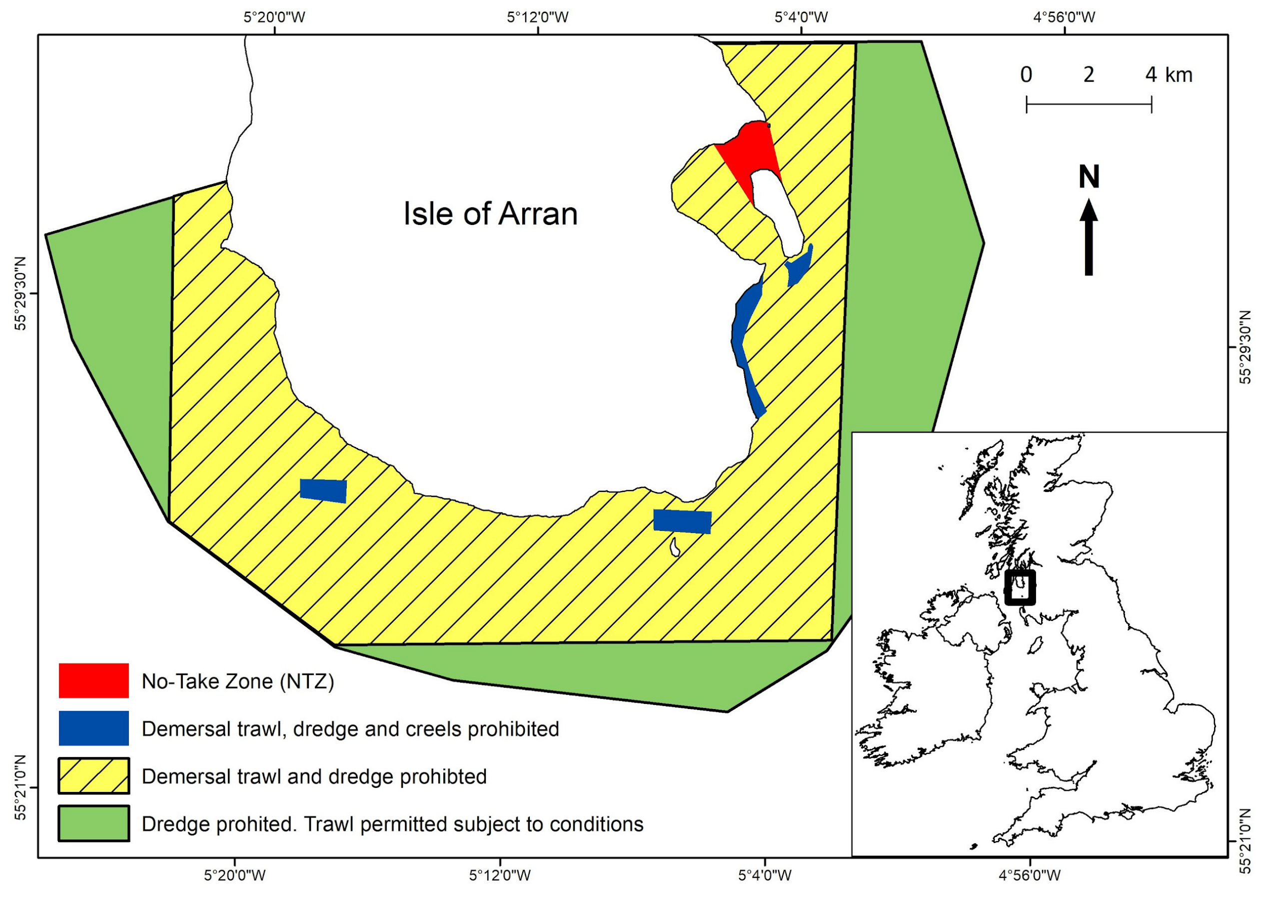

FIGURE 3 | Map of the south of the Isle of Arran and the various management measures inside the South Arran Nature Conservation MPA, as from February 2016. Inset shows the area in relation to the United Kingdom.

FIGURE 4 | The Story of COAST. **(A)** The three founding members at an outreach event in 2005 (Left to right: Don MacNeish, Howard Wood, Tom Vella Boyle); **(B)** COAST stall at the Arran Farmers Show, 2010 (Leigh Howarth in red coat); **(C)** Celebrating the designation of the Lamlash bay NTZ in 2008 (Left to right: Don MacNeish, Scottish Cabinet Secretary for Environment and Fisheries Richard Lochhead, Howard Wood, MSP Kenneth Gibson, Tom Vella-Boyle; **(D)** Howard Wood and Leigh Howarth doing the first SCUBA surveys of the Lamlash bay NTZ in 2010; **(E)** Surveying the South Arran MPA, 2014 (Left to right: Gus Robson, Howard Wood, Claire Youdale); **(F)** Celebrating the 10^{th} Anniversary of the Lamlash bay NTZ at the opening of the Octopus Centre, 2018. All identifiable persons in these figures have given consent for their images to be reproduced and used publicly.

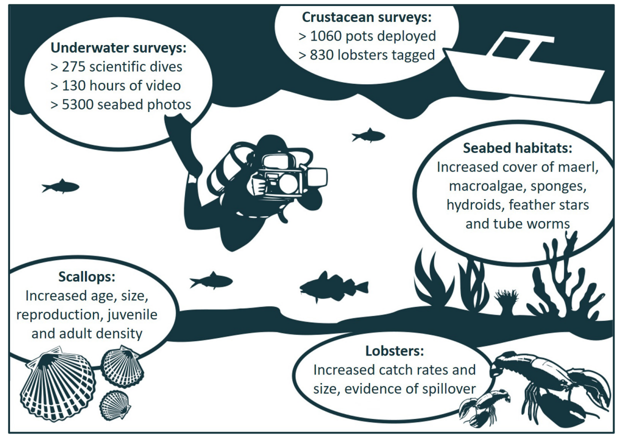

FIGURE 5 | Survey methods used in annual monitoring of Lamlash Bay NTZ where underwater surveys ran from 2010 to 2019 and crustacean surveys from 2012 to 2018. Also detailed is evidence of how protection has benefited seabed habitats and species of commercial interest. For full details, see Howarth et al. (2011, 2015a,b), Stewart and Howarth (2016), Crimmins (2018), and James (2019).

Committee (The Scottish Parliament, 2016). Although the Scottish Government (through Marine Scotland) is chiefly responsible for managing and enforcing the Arran and other Scottish MPAs (i.e., takes a top down approach), COAST continues to play an important collaborative role by advising on management measures, supporting and directing relevant science, and assisting with enforcement. For example, local residents, generally COAST members, have initially reported all of the relatively few known infringements in the NTZ and MPA around Arran, before the authorities have investigated them.

COAST's achievements in the field of marine conservation have not gone without notice. Environmental prizes to date include; the Observer Ethical Award in 2008, a place on the Scotsman Green List in 2009, the Nature of Scotland Award in 2014, the International Goldman prize for Europe and an OBE for services to the environment to Howard Wood in 2015, and the Spirit of the Community Award to COAST in 2017. The Goldman prize, considered the world's largest award honoring grassroots environmental activists[3] was particularly significant. Not only did it well and truly place COAST on the international stage, the prize money allowed the purchase of land and a building on the Lamlash bay foreshore to create the 'Octopus Centre'[4], officially opened in September 2018 to commemorate the 10^{th} anniversary of the Lamlash bay NTZ. This is Scotland's first MPA visitor centre, and the UK's first community-led MPA visitor centre. The Octopus Centre represents a new phase in COAST's history, providing a host of marine educational material and activities designed to educate and inspire both Arran residents and the increasing numbers of national and international visitors.

THE SCIENCE BEHIND COAST's SUCCESS

Monitoring of Lamlash Bay NTZ: 2010–Present

Routine monitoring of Lamlash Bay NTZ was initiated by the University of York in July 2010, 21 months after the NTZ was first established. These surveys were originally conducted by an MSc student and funded entirely by COAST (Howarth et al., 2011). Despite strong time and financial constraints, the project was a success thanks to substantial involvement from local volunteers and commercial fishermen operating within the area. Longer-term funding was then received from Fauna & Flora International (FFI[5]), the Blue Marine Foundation[6] and the Kilfinan Trust[7], all of which enabled the project to expand into a comprehensive

[3] www.goldmanprize.org/

[4] www.arrancoast.com/octopuscentre

[5] www.fauna-flora.org

[6] www.bluemarinefoundation.com

[7] https://bit.ly/2FFSNQB

FIGURE 6 | Juvenile queen scallops (*Aequipecten opercularis*) have settled on and attached to this kelp frond within the Lamlash Bay NTZ. Complex three-dimensional habitats such as these are essential for the early life history survival of many marine species, including commercially valuable species such as scallops and gadoid fish. Photo: Howard Wood.

annual monitoring program using a wide variety of survey methods (**Figure 5**).

Annual photo-quadrat surveys between 2011 and 2013 revealed a variety of seabed habitats, such as sponges and macroalgae, had become more abundant within Lamlash Bay NTZ than adjacent areas (Howarth et al., 2015a). Diver transects found that the recovery of these habitats had increased the abundance of juvenile scallops by two to five times compared to neighboring fishing grounds (Howarth et al., 2015b). Structurally complex 'nursery habitats' such as these are used by juvenile scallops (**Figure 6**) as a refuge from predation pressure (Howarth et al., 2011; Lambert et al., 2011). Hence, the Lamlash Bay NTZ appears to be promoting the recovery of nursery habitats, which in turn, is benefiting the recruitment of a commercially important species. In the long term, these effects are likely to increase the numbers of juvenile scallops entering the adult stock as a greater proportion of juveniles survive to reach maturity (Beukers-Stewart et al., 2003; Vause et al., 2007), although further monitoring is required to fully investigate this possibility.

Despite signs that protection had increased the abundance of juvenile scallops, surveys up to 2014 did not find a significant effect of the NTZ on the density of adult scallops (Howarth et al., 2015b; Boulcott et al., 2018). This lack of an early response in adult scallop density could be due to several factors including: (1) the small size of the NTZ; (2) a reduction in fishing pressure adjacent to the NTZ (on the control sites) after the NTZ was designated; (3) high variance of density estimates due to highly aggregated scallop distributions; and (4) episodic fluctuations in recruitment (Beukers-Stewart and Beukers-Stewart, 2009; Howarth et al., 2011, 2015b; Boulcott et al., 2018). Then again, both king scallop ages and sizes were significantly larger in the NTZ than outside (with differences increasing from 2010 to 2013), indicating that the population there was returning to a more natural, unfished state (Howarth et al., 2015b). Corresponding with these differences in body size, exploitable biomass (an indicator of commercial value) and reproductive biomass (an indicator of reproductive potential) were also greater within the NTZ than outside. Even more encouragingly, the most recent diver surveys, undertaken in July and August 2019, found that king scallop density in the NTZ has since increased dramatically, and is now more than 3.7 times higher than it was in 2013 (James, 2019). As a result, densities of adult scallops are now significantly higher in the NTZ than they are in an area still open to dredging off the NE coast of Arran (James, 2019). Overall, these findings indicate that the NTZ is protecting scallops from fishing mortality, allowing them to become more numerous, larger, older, and more fecund. This should mean the NTZ is contributing disproportionally to recruitment by exporting large amounts of eggs and larvae to surrounding areas (Beck et al., 2001; Gibb et al., 2007; Laurel et al., 2009; Harrison et al., 2012).

Annual crustacean surveys between 2012 and 2018 investigated the population dynamics of European lobster (*Homarus gammarus*), brown crab (*Cancer pagurus*) and velvet swimming crab (*Necora puber*) within the NTZ and directly outside its boundaries (Howarth et al., 2016; Crimmins, 2018). For most years, lobster catch rates were significantly higher within the NTZ, with differences being most dramatic for those above the minimum legal landing (e.g., 4.4 times higher in the NTZ than outside in 2018) (**Figure 7**). Lobster CPUE declined with increasing distance from the NTZ boundaries up to 20 km away (Howarth et al., 2016). Tagging and recapturing of the lobsters indicated this was likely due to 'spillover' with individuals from within the NTZ moving outside (Howarth et al., 2016; Crimmins, 2018). The body size of lobsters was also consistently greater within the NTZ across all years, and because egg production increases with body size (Cudney-Bueno et al., 2009; Harrison et al., 2012), and mature lobsters were so much more abundant in the NTZ, this difference translated to over 5.7 times more eggs within the 2.67 km^2 NTZ in 2018, than in an unprotected area of equal size. In combination, these results suggest that lobsters located within the NTZ are experiencing increased survivorship, allowing for establishment of higher densities, body sizes and greater reproductive output. In contrast, results for other crustacean species were more mixed. Catches of velvet crabs varied greatly from year to year with significant differences between inside and outside the NTZ for some years, but not for others. Catches of brown crab were consistently higher outside the NTZ than within. As brown crabs exhibited reverse trends to lobster, brown crabs may be being negatively affected through predation and/or competition by higher densities of large lobster within the NTZ (Howarth et al., 2016).

Research within Lamlash NTZ continues to provide invaluable evidence that temperate marine reserves can help to restore exploited stocks and the wider marine environment. Our surveys also highlight the importance of long-term monitoring to account for stochastic, annual fluctuations in abundance. They further demonstrate the importance of investigating multispecies interactions, as recovery of some species can have both positive and negative knock-on effects on others. Despite there now being over 350 MPAs in UK waters (JNCC, 2019), only one other, the Lyme Bay Marine Conservation Zone, has been studied in anything like the same detail as the Lamlash Bay NTZ (Mangi et al., 2011; Sheehan et al., 2013). Our results demonstrate that recovery of biological communities inside protected areas is not monotonic; instead, what we are seeing is complex,

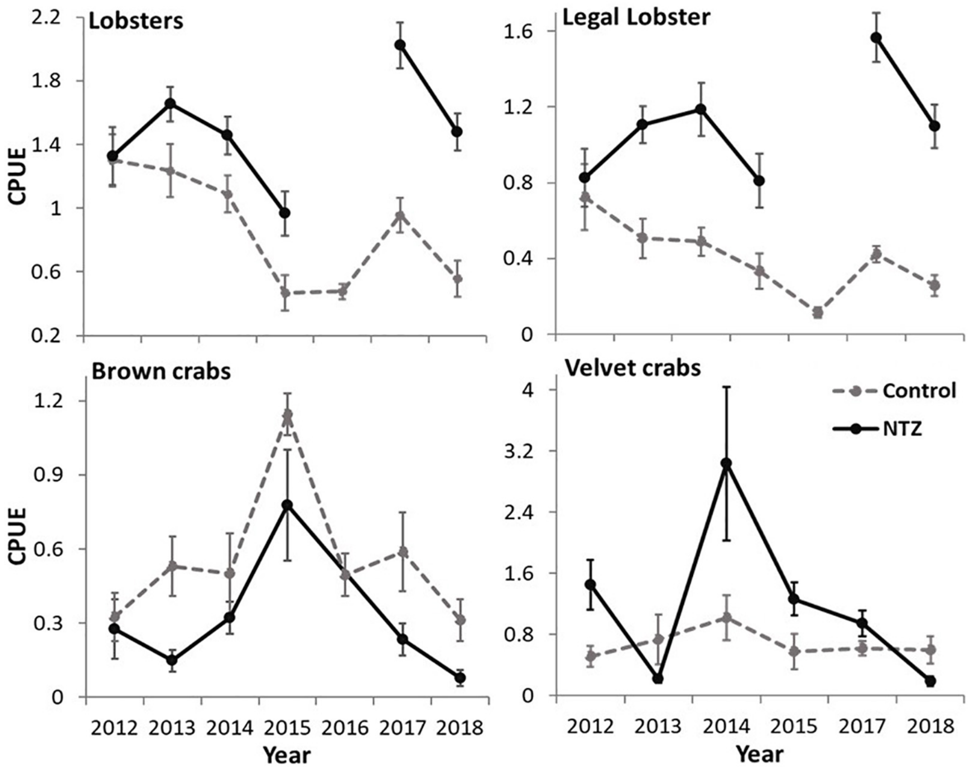

FIGURE 7 | Mean catch per unit effort CPUE (y-axis) of lobsters, legal-sized lobsters (> 88 mm carapace length), brown crab, and velvet swimming crabs within the NTZ and control sites in the surrounding fishing grounds over the 6-year study period (2012–2018). Black lines represents the NTZ and gray lines the control sites. Error bars represent ± 1 SE. There is no data for the NTZ in 2016 because it was not surveyed that year.

ecological processes unfolding in a dynamic environment. This should not be seen as problematic, the complexity should be embraced; it is a more accurate reflection of how ecosystems naturally function. This emerging understanding is crucial for both setting realistic management objectives for other MPAs in the region, and for managing the expectations of conservationists and managers in the future.

Making the Case for and Monitoring the South Arran MPA: 2011–Present

As stated earlier, COAST began building the scientific case for the larger South Arran MPA soon after the designation of the Lamlash Bay NTZ. The key first step was to identify the location and extent of 'priority search features' – vulnerable species and habitats recognized as such by national and international policies, such as seagrass, maerl, mussel beds, and sponge communities (COAST, 2012). Although considerable information was already available (e.g., on the seagrass beds in Whiting bay), there were known to be significant gaps in knowledge (COAST, 2012). From 2011 to 2012, COAST therefore undertook a large number of 'search dives' around the south of Arran, enlisting the help of local volunteers, Seasearch and University of York students. This information was then supplied to the Joint Nature Conservation Committee (JNCC), who provide UK governments with conservation advice, and validated by Scottish Natural Heritage (Howson and Steel, 2014), and was a key component in determining the extent of the proposed South Arran MPA and ultimately its successful designation (COAST, 2012).

Given the limited amount of quantitative biological data from Lamlash bay before designation of the NTZ, COAST were keen to avoid the same situation arising in the South Arran MPA. Management measures were scheduled to be implemented in the MPA in 2016, leaving the summers of 2014 and 2015 to conduct baseline surveys. Approximately, 40 dive surveys were done throughout the MPA over those 2 years (Hutchinson, 2015; Stark, 2015) using similar methodology to previous surveys (Howarth et al., 2011), but with dive transects also being videoed to provide a permanent record against which future surveys could be compared. Early analysis of dive surveys repeated at these sites in July and August 2019 suggests dramatic recovery after only 3.5 years of protection from dredging, with densities of king scallops over sixfold higher than in the previous surveys (James, 2019).

Researchers from the University of Glasgow, in collaboration with Scottish Natural Heritage, Marine Scotland Science, and recently the University of York, have also carried out a series of studies of fish nursery habitats within the South Arran MPA. These benefit from the mosaic of different seabed types within the MPA and the reduced disturbance in later years once trawling and dredging had been restricted (David Bailey, personal communication). These studies have revealed differences in nursery habitat use and other behaviors in the three main commercial gadoid fish species (cod, haddock, and whiting) (Elliott et al., 2017b, 2018) with evidence that cod in particular responds positively to higher benthic biodiversity and landscape heterogeneity (Elliott et al., 2017a,b). This work, which has involved over 600 camera drops at the time of this publication, provides a dataset covering the designation and protection of the wider MPA, which will allow changes to its protected features, biodiversity and ecosystem functions to be determined. The protected ground of the MPA will become an important area for the study of marine landscape ecology, a topic with great potential to improve our understanding of marine ecology and to improve management practices.

Finally, the role of the South Arran MPA for storing carbon, and hence helping to mitigate climate change, is now also being to be recognized. In a recent assessment of the contribution of Scotland's MPAs to carbon storage (Burrows et al., 2017), the South Arran MPA was assessed as providing the fifth highest contribution, with an estimated stock of total carbon (organic and inorganic forms from both biological and geological (sediment) sources) of 8,046 tons per km^2. Burrowed mud, which makes up a significant proportion of the South MPA ($\sim$ 160 km^2) is known to be a particularly important long-term carbon store, especially when it is not disturbed by trawling or dredging (Smeaton and Austin, 2019).

BENEFITS BEYOND BOUNDARIES

COAST's influence on community-led marine protection goes well-beyond the shores of Arran. Whilst COAST's founders set out with a very specific goal – to close off and manage a small area of the sea for natural regeneration – their activities have sent waves of change around the UK and beyond.

Public Outreach

A core part of COAST's mission since its formation has been to undertake public outreach in order to educate people about both marine conservation issues and to build support for potential solutions. This has included running regular public activities on Arran (e.g., rock pool rambles for the public, school visits, stalls and displays at community events) and now hosting visitors at their Octopus Centre (over 11,000 between August 2018 and August 2019). Media engagement has also been used to build public and political support since the beginning of COAST's campaign for the Lamlash bay NTZ. One of the most important productions was the near hour long 2006 documentary about COAST called 'Caught in Time' filmed and produced by Doug Anderson, with strong Arran connections and now a renowned BBC wildlife cameraman: www.youtube.com/watch?v= fdAJK7CkNbQ&feature= youtu.be. This came at a time when their campaign appeared to be stalling, despite widespread local and scientific support. A further 25 min of live coverage on BBC News 24 in May 2007 continued to raise public awareness, the year before the NTZ was finally designated. More recent print and online highlights include the Times in July 2018 (https://bit.ly/2U2ED6c); the Smithsonian Institution in January 2017 (https://s.si.edu/2HUH8QJ); the New York Times in August 2015 (https://nyti.ms/2JNRft5) and National Geographic in April 2015 (https://bit.ly/2I52IBQ). There has also been considerable national television coverage, most recently features on the UK's BBC – Blue Planet UK in March 2019, Countryfile in February 2019, and Springwatch in June 2018, and internationally on Al Jazeera in 2013 (https://bit.ly/2uzSj9F). In 2010, the Franco-German TV channel Arte also made a 10 min documentary for distribution across Europe, particularly France and Germany, but also the Netherlands, Belgium, Luxembourg, Switzerland, Austria, Italy, and Israel.

Scientific outputs from the Universities of York and Glasgow on the research done on the Lamlash Bay NTZ and South Arran MPA have also been widely promoted through the media, generally in collaboration with COAST. These have again attracted widespread national and international coverage (e.g., The Scotsman https://bit.ly/2uABOdq and https://bit.ly/2FFuL8n; Futurity https://bit.ly/2uydp82 and https://bit.ly/2FInqWC), and helped promote both the Isle of Arran marine conservation success story, and the supporting scientific research.

Scientific Influence

The Universities of York and Glasgow have published 10 peer-reviewed journal articles, 1 report and 1 book chapter on the marine ecosystems around Arran and in the Clyde, since 2010. Collectively, these publications have been cited 274 times (Google Scholar 20/01/2020). This research helps fill an important knowledge gap because the recovery of marine ecosystems inside MPAs in temperate waters remains understudied (Fenberg et al., 2012). In terms of community led MPAs in temperate waters, the work is even more unique. COAST have also played an important role in nurturing young marine scientists. To date, they have hosted 24 BSc, MSc, and Ph.D. research projects from the Universities of York, Glasgow, Heriot Watt, Edinburgh Napier, and St Andrews, Bangor University, University of the West of England, and University College London. Furthermore, 17 young scientists and conservationists have traveled to Arran to work with COAST, from a diverse range of countries such as the UK, the Irish Republic, Germany, Greece, Canada, New Zealand, and South Africa.

Supporting Other Marine Conservation Campaigns

COAST's success with the Lamlash Bay NTZ not only helped support their campaign for a much larger MPA designation around the south of Arran, it has also kick-started and inspired a national movement which has empowered and united other communities and increased grassroots participation in marine management.

A significant early step was helping to form the Sustainable Inshore Fisheries Trust (SIFT www.sift-uk.org/) in 2011, with Howard Wood from COAST as a founding board member. SIFT's stated goal is '*promoting the sustainable management of Scotland's inshore waters so that they provide the maximum long term socio-economic and environmental benefits to all Scotland's coastal communities.*' SIFT therefore has a broad remit, but its initial focus was guided by the concerns of COAST and others about the degraded state of the Clyde. Following almost 3 years of research and stakeholder engagement from 2013 to 2016, SIFT submitted an innovative spatial management plan to the Scottish Government, aimed at helping revitalize Clyde fisheries and marine ecosystems. Although not ultimately taken forward by the Scottish government, their work has kept the Scottish government focused on the plight of the Clyde and many of their ideas have been adopted by the Clyde Marine Planning Partnership (see earlier). SIFT has had more success recently, helping to run a campaign which stopped a large-scale commercial proposal by a biopolymer company to dredge for kelp in Scottish waters.

Since 2014, COAST has also been working with Fauna & Flora International (FFI), and a range of partners in Scotland, to support coastal communities in playing an active role in marine conservation. The need for this collaboration was borne from a multitude of requests being brought to COAST from other aspiring communities and individuals – keen to replicate the success of COAST and enhance the protection of their own local waters.

Local people living and working along the coasts of Scotland have a unique dependence on marine resources, a wealth of knowledge and skills, and specific aspirations for reforms within traditional marine management decisions. Through supporting local community groups in having a stronger voice in support of marine protection, COAST and FFI have supported the emergence of new community organizations such as the Community Association of Lochs and Sounds (CAOLAS[8]) based around the shores of Loch Sunart and the Sound of Mull. The partnership has also influenced the achievement of specific conservation milestones including the establishment of Scotland's first Demonstration and Research MPA in Fair Isle, the increased exclusion of bottom-towed fisheries within MPA management measures, and the formation of a community-driven, national Coastal Communities Network[9] in Scotland. The emerging network is already larger and more influential than anticipated, involving 16 community groups at present, with strong indications that this will continue to grow. This new platform for local communities to engage in Scottish marine management has not only increased participation at the grassroots level, but it has also provided a key mechanism for government agencies such as Marine Scotland and Scottish Natural Heritage to engage more effectively with communities. This is in line with Scotland's evolving Community Empowerment agenda and more akin to the co-management frameworks that are seen in other parts of the world.

Outside of Scotland, COAST has had significant influence, delivering talks and advice across the UK, America, and Europe. The latter through participation in the annual Europarcs conference[10]. Connections have been made with those setting up NTZs in Mauritius, and in linking together coastal communities networks in Tanzania. Greenpeace also chose the Lamlash Bay NTZ as one of three MPAs in the world to demonstrate 'Why ocean sanctuaries are so important,' as part of their 2018 campaign for the world's largest ocean sanctuary in the Weddell Sea, now signed by almost 2.8 million people: https://bit.ly/2HPhkCS; www.facebook.com/greenpeaceuk/videos/10155626965768300/?v = 10155626965768300.

Wider Policy Impact

Conclusively demonstrating wider policy impact is difficult because government reports and policy documents rarely reference specific case studies or scientific publications. However, we have provided strong evidence above to highlight the significance of COAST's efforts to improve marine conservation from a local to international level. At a national level, COAST have recently set up the 'MPA Management Plan Project.' This project is using marine science, law and socio-economics to produce a bottom–up model of MPA governance and management, which aims to transform policy and actions of government and regulatory bodies to establish an effective, sustainable and adaptive MPA management system. COAST will lead on and promote a culture of best practice by establishing an effective model for managing the South Arran MPA, which is shared across the Scottish MPA network and communities. The legal governance component of this project is complete and four briefing papers have been produced in partnership with Edinburgh Law School[11].

Unfortunately, within the UK more widely there is still a distinct lack of MPAs offering the same high level of protection provided to the Lamlash bay NTZ, or even the South Arran MPA. In fact, a recent Parliamentary enquiry by the UK's Environmental Audit Committee (House of Commons, 2017) stated that without fully protected reference areas "*the Government will be unable to establish an effective and coherent MPA network, as they will have no benchmark against which to assess the effectiveness of management measures*." The recovery of marine life inside the Lamlash Bay NTZ was highlighted by a number of witnesses at this enquiry as being a clear demonstration of what is possible with both community support and a high level of protection. The generally inadequate management of UK MPAs at present reflects global patterns, however, now is not the time to ignore the issue, rather it is the time to further study and promote the few areas of the seas which are fully protected to better inform management elsewhere.

LESSONS LEARNED AND THE FUTURE

The Lamlash Bay NTZ and South Arran MPA join a select group of MPAs that have punched well-above their weight. The others such as Leigh marine reserve in New Zealand (Ballantine, 2014), Apo Island in the Philippines (Russ and Alcala, 1999) and Las Cruces in Chile (Navarrete et al., 2010),

[8] www.caolas.org
[9] www.communitiesforseas.scot
[10] https://bit.ly/2OB0Emr
[11] www.law.ed.ac.uk/research/research-projects/saving-our-seas-through-law

are similarly small, but have been highly influential. They also have in common that they have been passionately fought for and thoroughly studied. On Arran, the campaign for better protection of their seas was kick-started and taken forward by a small band of committed and inspirational leaders, but its success equally lies with the engagement and support from the local community (Sutton and Rudd, 2016). This was not only crucial for building the case that the seas are a public good which should be managed for the benefit of all, but it also made COAST more resilient to the dynamics and challenges of a long, at times fraught, and ongoing campaign. Approaching and building relationships with legal experts, civil servants and politicians from the local to national level was vital for understanding the legislative frameworks and political systems that their campaigns have had to negotiate. Furthermore, COAST have skilfully used the media to promote their aspirations and to build support for their cause well-beyond the shores of Arran. Finally, COAST engaged with scientists early in their campaign and built up mutually beneficial relationships. Appropriate and timely science can inform, support and guide campaigning too much greater effect than is possible in its absence. Science therefore produces a solid platform for informed campaigning, enabling campaigners to speak with authority and for their arguments to be more resistant to inevitable attacks. With this recipe of ingredients COAST were able disrupt the *status quo* of the marine management system in Scotland that at the time was at best ambivalent, at worst resistant, to change, and in doing so inspire other local communities to take the destiny of their coastal waters into their own hands.

COAST continues to move forwards. With their Octopus Centre, they are playing a growing role in marine education and outreach, but they are also helping mentor other coastal community groups more than ever before. Management of UK seas is improving, but still has a long way to go before we will truly have an ecologically coherent network of MPAs. On the policy front, UK marine management faces further challenges due to the UK leaving the EU (Brexit; Solandt et al., 2017; Stewart and O'Leary, 2017; Stewart et al., 2019) and of course, anthropogenic ocean warming and acidification are increasingly affecting the marine ecosystems themselves. Howard Wood, one of the founders of COAST, has now become a Global Climate Action summit 'Climate Trailblazer'[12], in order to turn his hand to these even bigger issues. Nevertheless, COAST's most important message is that any environment can benefit from better protection, and that every community has the right to a better environment if they want one. If that is embraced on a global scale then we truly will see a Seachange in the management of our seas.

ETHICS STATEMENT

All field and laboratory work in this study received ethical approval from the Department of Environment and Geography, University of York. Sampling of specimens in the No Take Zone was done under permit from Marine Scotland.

AUTHOR CONTRIBUTIONS

BS and LH jointly coordinated this effort, led the writing, and conducted much of the science. HW provided most of the information and photos about the story of COAST. KW provided information about the wider influence of COAST. WC and ÉC provided the data and the figure on crustaceans. JH and CR helped to supervise the research and obtain the funding. All authors assisted with the writing of the manuscript.

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There is a large array of people to thank for contributing to this unfolding story. For being there with the vision and commitment from the start, none of this would have been possible without the two other founding members of COAST, Don MacNeish and Tom Vella-Boyle. Likewise, the members of COAST and the numerous staff and volunteers who have worked tirelessly, often for many years, have been equally important. The two directors of COAST, Andrew Binnie, who led the proposal for the South Arran MPA, and Paul Chandler, who has overseen the opening of the Octopus Centre, also deserve special mention. Russell Cheshire, the current chair of COAST, is another long-standing supporter who has provided essential assistance with the scientific fieldwork. The support of local fishermen and boat owners, Charlie Weir, Ian Cusick, Tim James, Gavin Cameron, and Jason Coles has been crucial for running the crustacean and diving surveys since 2012. Many students from the University of York helped to collect and analyze the data reported here, particularly Tim Cross, Pascal Dubois, Lowri Evans, Daniel Steadman, Sarah Pickup, Paul Gratton, Matthew Judge, Will Hutchinson, Jenny Stark, Brian Christie, James Spilsbury, Lauren James, and William Notley. We further thank Dr. David Bailey, for providing information on the baited remote underwater video survey work done by Glasgow University. We gratefully acknowledge the generous financial support from the Esmée Fairbairn Foundation, Fauna & Flora International, the Kilfinan Trust, and the Blue Marine Foundation to both COAST and the University of York, which made the work presented here possible.

[12] www.globalclimateactionsummit.org/trailblazers/

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A General Methodology for Beached Oil Spill Hazard Mapping

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The current lack of a standardized approach to compute the coastal oil spill hazard due to maritime traffic accidental releases has hindered an accurate estimate of its global impact, which is paramount to manage and intercompare the associated risks. We propose here a hazard estimation approach that is based on ensemble simulations and the extraction of the relevant distributions. We demonstrate that both open ocean and beached oil concentration distributions fit a Weibull curve, a two-parameter fat-tail probability distribution function. The simulation experiments are carried out in three different areas of the northern Atlantic. An indicator that quantify the coastal oil spill hazard is proposed and applied to the study areas.

Keywords: oil spill modeling, MEDSLIK-II, oil spill hazard mapping, Weibull pdf, hazard index

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1. INTRODUCTION

Risk assessments have been widely employed by scientists and engineers as a tool to deal with harmful natural events (e.g., earthquakes) and anthropogenic events (e.g., oil spills) that we are unable to describe and/or predict with fully deterministic methods. A critical step within a risk assessment framework (e.g., ISO 31000), hazard quantification (or hazard mapping) is the probabilistic estimation on the frequency of occurrence and magnitude, based on historical observations of the events of interest. The seminal paper by Gutenberg and Richter (1944) clearly illustrates how the careful analysis of the probability distribution function (PDF) of seismic events, i.e., the probability of observing an earthquake of a certain magnitude in a limited area over time, led to an objective hazard quantification method and triggered comparisons within different tectonic areas around the globe (Shedlock et al., 2000). This well-established seismic hazard framework suggests that an appropriate statistical description of the variable of interest is of fundamental importance to classify hazard and then risk. As far as the authors are aware, there has been no study dedicated to the description of beached (or floating) oil concentration PDF, based on either simulated or observed data.

The oil spill hazard assessment framework lags behind these well-established hazard assessment fields. There is a severe lack of observational data, especially oil concentrations along coasts after spill events (in the paper also referred to as beached oil concentrations), eliminating the possibility of empirically describing the coastal oil hazard. Numerical ensemble oil spill modeling has been widely used to tackle the data gap, emulating reality, i.e., spill fate and generating information for hazard quantification (e.g., Price et al., 2003; Liubartseva et al., 2015; Sepp Neves et al., 2015; French-McCay et al., 2018). However, there is currently no standard way to describe the beached oil concentrations obtained by the ensemble experiments and, therefore, mapping the relative coastal hazard. Consequently, we are still unable to compare different hazard estimates, to evaluate objectively the completeness of the ensemble datasets and, most importantly, to be able to establish a global coastal oil spill hazard assessment framework.

The estimation of the probability distribution functions of atmospheric state variables and tracer/contaminant concentrations has attracted the attention of meteorologists for decades. It has long been recognized that wind speed PDFs fit a Weibull distribution (Justus et al., 1976; Pavia and O'Brien, 1986; Monahan, 2006), characterized by a large number of low magnitude events (e.g., low wind speed), large variance and, most interestingly, "anomalously" frequent high magnitude events, typical of fat-tailed distributions. Observational and modeling experiments (Tirabassi et al., 1991a,b) have shown that wind speed plays a major role in shaping tracer concentration distributions, which were also found to fit a Weibull PDF. In the ocean, recent papers have documented that surface coastal currents (Ashkenazy and Gildor, 2011), i.e., Gulf of Eilat/Aqaba, and global ocean currents (Chu, 2009) also fit a Weibull distribution, suggesting that ocean pollutants could also be similarly distributed as observed in the atmosphere.

The linkage between flow field characteristics and tracer concentration PDFs has been described by, among others, Hu and Pierrehumbert (2001) for stratospheric flows. They explain that every flow feature (e.g., eddies, filaments) has its own characteristics such as its shape and length, speed and direction, and how long it will last before disappearing/merging with other structures. Tracer concentration PDFs are primarily a result of different "parcels separation rates" taking place in the different parts of the concentration field, modulated by the flow field features they are exposed to. Areas with a relatively complex circulation are expected to present both features with lower (e.g., western boundary currents) and higher (e.g., filaments or eddies) "separation rates." In other words, as the flow field becomes complex and rich in features, the tracer concentration PDF increasingly depart from a Gaussian distribution. Therefore, we expect that if ocean feature-rich current patterns are Weibull distributed, the marine tracers concentrations, including beached oil, should be similarly distributed.

Recently Alves et al. (2016b) gave evidence that ocean current conditions determine the fate of the oil at the coasts. Currents are by themselves affected by bathymetry and they contain the information about the specific flow field that is consistent with the coastline constraint and the other forcings. In this paper we add to these findings by estimating the PDF of accidental oil releases, both at sea and along the coasts. Our definition is the one of "standard" accidental oil releases, corresponding to a volume of oil spilt equal to 10,000 tons, as pointed out but the analysis of Huijer (2005). Firstly, we will show that at the surface and in the open ocean, far from coastal boundaries, the oil concentration fits a Weibull PDF. Secondly we will simulate an ensemble of oil releases near the coasts and we will analyze the PDF of the beached oil. Three very different Atlantic coastline segments and ocean current regime areas will be considered for different current and wind conditions during the year 2013. A Weibull PDF function is estimated from the data and a coastal oil spill hazard index will be proposed in terms of the "Weibull tail distribution."

After a description of the framework for oil spill hazard (section 2) and the ensemble experiments performed (section 3), the statistical distribution of oil concentrations in surface waters is analyzed (section 4). In section 5 we will then analyze the PDF for the beached oil on three coastlines around the Atlantic basin. Section 6 will offer the analysis of a new hazard indicator extracted from the statistical distributions of the beached oil in the three areas. Final overview of the results and future work is presented in section 7.

2. STANDARDIZED APPROACH TO EVALUATE OIL SPILL HAZARD

The framework for oil spill hazard and risk assessment has been defined by adapting the generic and recognized standard, i.e., ISO 31000, to oil spill model simulations so that the oil spill hazard assessment could be carried out in a scientific and standard way (Sepp Neves et al., 2015). The authors identify ensemble oil spill simulations as the method to produce the basic data to estimate the hazard.

Later, the technique of ensemble oil simulations was extensively analyzed for one coastal stretch, the Algarve coastal area in Portugal (Sepp Neves et al., 2016). The ensemble simulations were carried out using an extensive identification of modeling uncertainty sources: the meteo-oceanographic fields used for the oil spill simulations (different wind and ocean models were used), the inclusion of the Stokes drift, the oil spill release points, the oil spill model setup in terms of volume of oil spilled, time of spillage, and duration of the spill. Ten different model configurations were used to obtain the hazard of the beached oil at the Algarve coasts. The paper concludes that the beached oil distribution is non-Gaussian so that a hazard risk index which would use mean and standard deviations is not appropriate. In the same paper, Sepp Neves et al. (2016) show also a probability distribution of beached events that is analog of the pdf discussed and revealed in this paper (see Figure 10 of Sepp Neves et al., 2016). However, the specific PDF was not identified, and the hazard index not clarified.

In this paper we extend the simulations to 3 different coastal areas and to the open ocean, trying to show the generic character of the pdf resulting from the oil spill simulations as defined in Sepp Neves et al. (2015), Sepp Neves et al. (2016). In this paper we choose only to sample the uncertainties connected with the oil release points and the meteo-oceanographic conditions, together with the structure of the coastline. We consider operational volumes of oil spilt equal for all three areas. As shown in Sepp Neves et al. (2016) the inclusion of Stokes drift does not change the structure of the resulting beached oil PDF so we have not included the uncertainty connected to the Stokes drift.

The methodology of oil spill hazard mapping requires an extensive number of simulations due to the large number of uncertainties connected with oil spill conditions and model parameters. One principle of importance is the statistical independence of each release point simulation with respect to meteo oceanographic conditions that are responsible for the transformation and transport of the oil. In Sepp Neves et al. (2016) and in this paper we use the spatial and temporal decorrelation time of ocean mesoscale eddies as the threshold for successive simulations, but other criteria might be used if higher frequency ocean processes, such as tides, will be considered in the future. Furthermore, the length of the simulation should be a good compromise with ocean currents statistical independence and weathering processes time scale that has been estimated to be of the order of few days for similar amount of oil and duration of the spill (Coppini et al., 2011).

In this paper we would like to show that, independently from the coastal geometry, the beached oil PDF has the same structure, changing its parameters in dependence of the flow field characteristics and the wind induced transformations. The three areas considered in this study have very different mean current regimes: the Eastern Atlantic Archipelago is characterized by a relatively weak and broad southwestward flow, the Western Atlantic Island is characterized by the intense zonal Caribbean current flow and the Basilian coastal stretch is characterized by a western boundary current regime. Given these differences, the three areas differ greatly for the spatial scales and intermittency of the mesoscale eddy field, as well as the wind regimes.

TABLE 1 | Configuration of the ensemble experiments for the three areas of the Eastern Atlantic archipelago, the Western Atlantic Island and the Brazilian state of Bahia.

Experiment configuration variable	Value
Volume of oil (tons)	10,000
Type of oil (API)	38
Duration of the spill (h)	48
Currents	CMEMS (*)
Winds	ERA-Interim (**)
Coastline resolution	∼1 km
Coastal type	Sand
Length of simulation	240 h

(*) http://www.marine.copernicus.eu - GLOBAL ANALYSIS FORECAST PHY 001 024, the product is released by Mercator Ocean International. (**) ERA-Interim are released by ECMWF.

3. ENSEMBLE EXPERIMENT SETUP

Ensemble oil spill simulations have been used in the past to assess hazard at the coasts and in the open sea (Price et al., 2003; Liubartseva et al., 2015; Sepp Neves et al., 2015; Olita et al., 2019) but surprisingly not much has been done with respect to the derived oil concentration frequency distributions. Here we use an ensemble simulation approach to define the PDFs for oil spill release events occurring near the coasts.

Our ensemble methodology consists of varying the oil release positions, changing the met-ocean conditions and collecting the concentration of oil either in the open ocean or along the coasts. This methodology allows to sample the uncertainty in hazard mapping due to the unknown location of the spill and different meteo-oceanographic conditions. Hazard mapping deriving from a sufficiently large number of met-ocean conditions and release points is then used to describe the PDF of oil pollution events at the coasts.

The oil spill transport and transformation model is MEDSLIK-II that solves an advection-diffusion equation for the oil concentration and its transformation processes (De Dominicis et al., 2013). The transformation of oil (or weathering) considers evaporation, emulsification, dispersion, and spreading processes connected to the slick. The advection-diffusion part of the equation is solved by a series of lagrangian equations for the position increments, $d\vec{x}$, of the parcels that compose the slick at the surface, e.g.,

$$d\vec{x} = U(\vec{x}, t)dt + \sqrt{2\kappa\, dt}\vec{z}(t) \tag{1}$$

where $\vec{x} = (x, y)$ is the horizontal position of the parcels, dt is the time increment, $U(\vec{x}, t)$ is the two dimensional ocean transport field, κ is a constant horizontal turbulent diffusivity value and $\vec{z}(t)$ are independent random numbers normally distributed, parameterizing lagrangian diffusion. The model considers only surface releases, the slick is evolved considering four type of parcels: surface, dispersed, sedimented and beached parcels. For the surface parcels, Equation (1) is used with a prescribed κ (taken to be equal to 2 m^2s^{-1} and current velocity analysis fields, as described in **Table 1**. Earlier studies (Dominicis et al., 2014) of lagrangian diffusivities found $\kappa \sim 1 - 100\ m^2s^{-1}$ and here we have chosen the lowest value possible in order to diminish the importance of diffusion with respect to advection by the currents. The dispersed particles are generated by the oil transformation processes (De Dominicis et al., 2013), they evolve separately from the surface oil parcels and they compose a volume of submerged oil that we do not consider in this paper except if the dispersed oil becomes beached oil. In MEDSLIK-II oil transformation processes are associated only to density, expressed as American Petroleum Institute (API) gravity.

Oil parcels arriving at the coasts become beached oil parcels when their trajectories cross pre-defined coastal segments. The permanent beaching and the probability of wash-off will depend on the coastal type, as explained in De Dominicis et al. (2013), however the coastal oil holding capacity is the same for all types of coasts and it is currently set to 5,000 bbl/km. In our analysis we use the total beached oil, the sum of the permanent and re-detachable oil. The simplistic beaching model is in accordance with the limited temporal and spatial resolution of the employed meteo-oceanographic models and in the past it has generated satisfactory results in real spill cases at similar scales (Coppini et al., 2011).

All experiments are carried out supposing several Release Points (RP) off the coastlines (**Figure 1**). The RP grid encompasses the main maritime routes simulating potential oil releases from the maritime corridors. Given the amount of oil released, 10^4 tons (**Table 1**), our simulations encompass the release from potential accidents (Huijer, 2005). The current field $U(\vec{x}, t)$ used in (1) is derived from the surface analysis fields of the Copernicus Marine Environment Service (CMEMS) at 1/12° spatial resolution (Lellouche et al., 2013)[1]. The atmospheric winds and air temperature used in the weathering processes are derived from the European Center for Medium range Weather

[1]http://marine.copernicus.eu/ – GLOBAL ANALYSIS FORECAST PHY 001 024, the product is released by Mercator Ocean International.

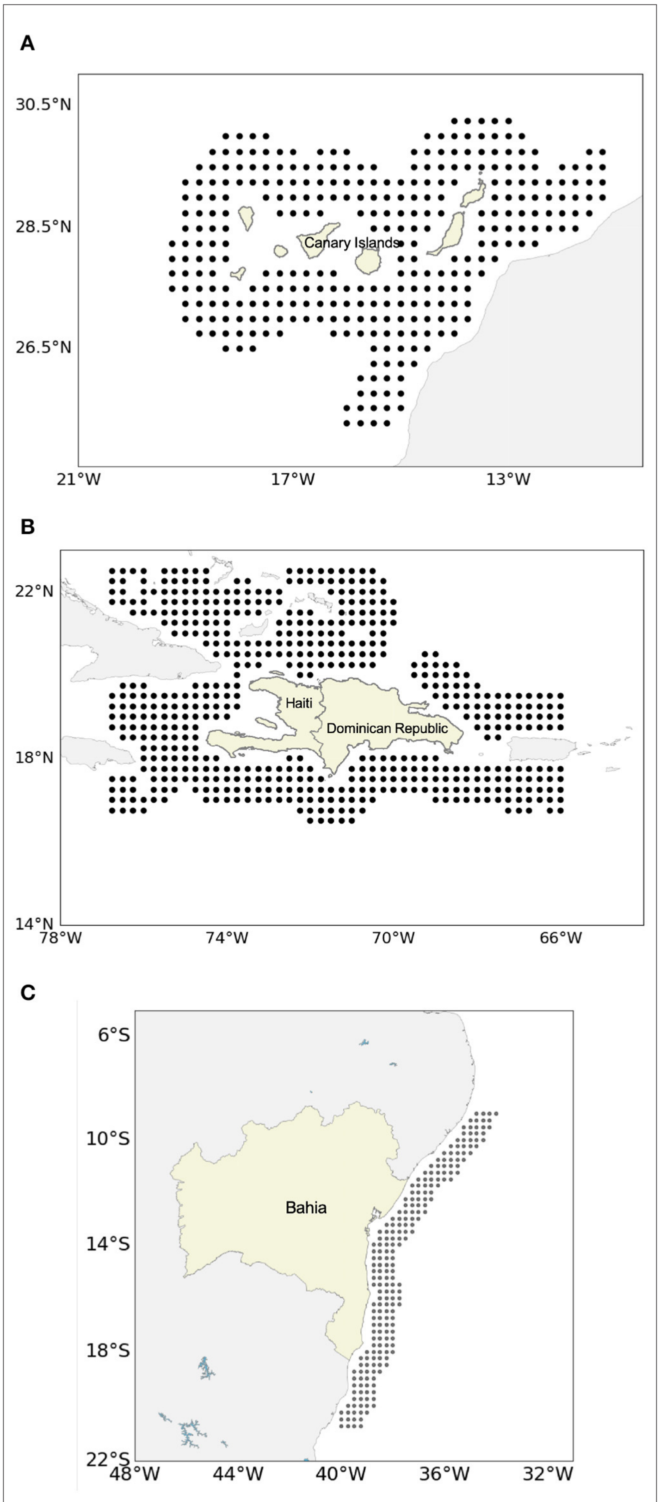

FIGURE 1 | Map of release points (black dots) for the **(A)** Eastern Atlantic Archipelago, **(B)** the Western Atlantic Island, and **(C)** coastline of Bahia (Brazil).

Forecast (ECMWF) ERA-Interim at 1/8° (Dee et al., 2011). The input winds are given every 6 h and linearly interpolated to the oil spill model time step which is 1 h.

For each release point, 240 h-long oil spill simulations were launched every 10 days throughout year 2013, using the experiment configuration described in **Table 1** for three different areas in the Atlantic ocean. All different areas used the same wind and ocean current data set, the same spilled volume, type and duration of the spill (see **Table 1**). The coastline segments were derived from the global Wessel and Smith (1996) data set which represent the coastlines with polygons of a few hundred meters resolution for the world ocean. Coastal segments were considered at 1 km resolution for the three study areas because this is recommended if details of the coastline are not known from complementary *in situ* data.

The statistical independence of each release point simulation, i.e., the occurrence of one beached oil event not affecting the probability of occurrence of the other due to the same release point, was preserved in two ways. Firstly, the time interval between consecutive simulations from a single release point was longer than the typical Lagrangian decorrelation time in the Atlantic ocean (∼5 days, see Maximenko et al., 2012). Secondly, the distance between neighboring release points (i.e., 25 km) was chosen based on estimation of the mesoscale spatial decorrelation scale from satellite altimetry that is at its minimum 50 km (Le Traon et al., 1990). The 10 days long simulations were chosen as a good compromise between this statistical independence and the transformation processes occurring at the surface that will evaporate and disperse the oil in the first few days for the amount and modalities of oil released in the simulations (2 days release time of 10,000 tons), as shown for example in Coppini et al. (2011).

4. THE OPEN OCEAN SURFACE OIL CONCENTRATIONS PDF

The first aim of the paper is to show the PDF of surface oil in the open ocean, due to a single oil spill event repeated every 10 days for the entire year 2013. A single release point is chosen off the Portuguese coast, located in the open ocean flow field (**Figure 2A**) using the model configuration described in **Table 1**. The experiment considers a "virtual" coastline, *sampling line*, defined 50 km southward of this release point, where the oil concentration is monitored to study its statistical distribution due to the oil advective-diffusive dynamics with and without the oil transformation processes (weathering and no-weathering experiments). All non-zero surface oil concentration values observed along the sampling line grid cells are stored every 6 h throughout the duration of each simulation.

In total, weathering and no-weathering experiments generated 3,779 non-zero oil concentration values each along the sampling line grid boxes and their concentration histograms are presented in **Figure 2**. Both histograms show that most of the events accumulated in low concentration bins but that in general, the range of concentrations observed is large, spanning two orders of magnitude for the oil concentration. Very high concentration events were found to be present, depicting a fat-tailed histogram.

Based on the histogram characteristics we choose a Weibull distribution function, $W(x;\beta,\eta)$ for a maximum likelihood estimation of the PDF:

$$W(x;\beta,\eta) = \frac{\beta}{\eta}\left(\frac{x}{\eta}\right)^{\beta-1}\exp\left(-\frac{x}{\eta}\right)^{\beta} \tag{2}$$

where x is the oil concentration, β and η are the shape and scale parameters respectively. In the Weibull function (2), the shape parameter is the one that controls how different from a Gaussian distribution is the PDF. For β less than 1 the distribution has its maximum at values close to zero. For $\beta = 1$ it is an exponential and for β greater than 1 the distribution has a single maximum for oil concentrations for relative large positive values and exhibits a marked asymmetry.

The Weibull distribution function has a wide range of applicability and it was in fact invented by Weibull and Stockholm (1951) to describe the failure of mechanical systems under various types of stresses or the size distribution of fly ash. Here we show for the first time that the surface oil in the ocean fits the same distribution as the currents (Chu, 2009; Ashkenazy and Gildor, 2011). It is interesting also to note that the Langevin-type equations, like the ones used in our oil spill modeling equations (1), produce probability distribution functions that are skewed and heavy-tailed (Sardeshmukh and Penland, 2015). Thus it

TABLE 2 | Number of Release Points (RPs), simulations and beached oil events for year 2013 for the the three different areas.

Parameters	Eastern Atlantic Archipelago	Western Atlantic Island	Bahia (Brazil)
Number of RPs	372	317	432
Number of simulations	13,764	11,729	15,984
Number of beached oil events	4,617 events	11,420 events	4,129 events

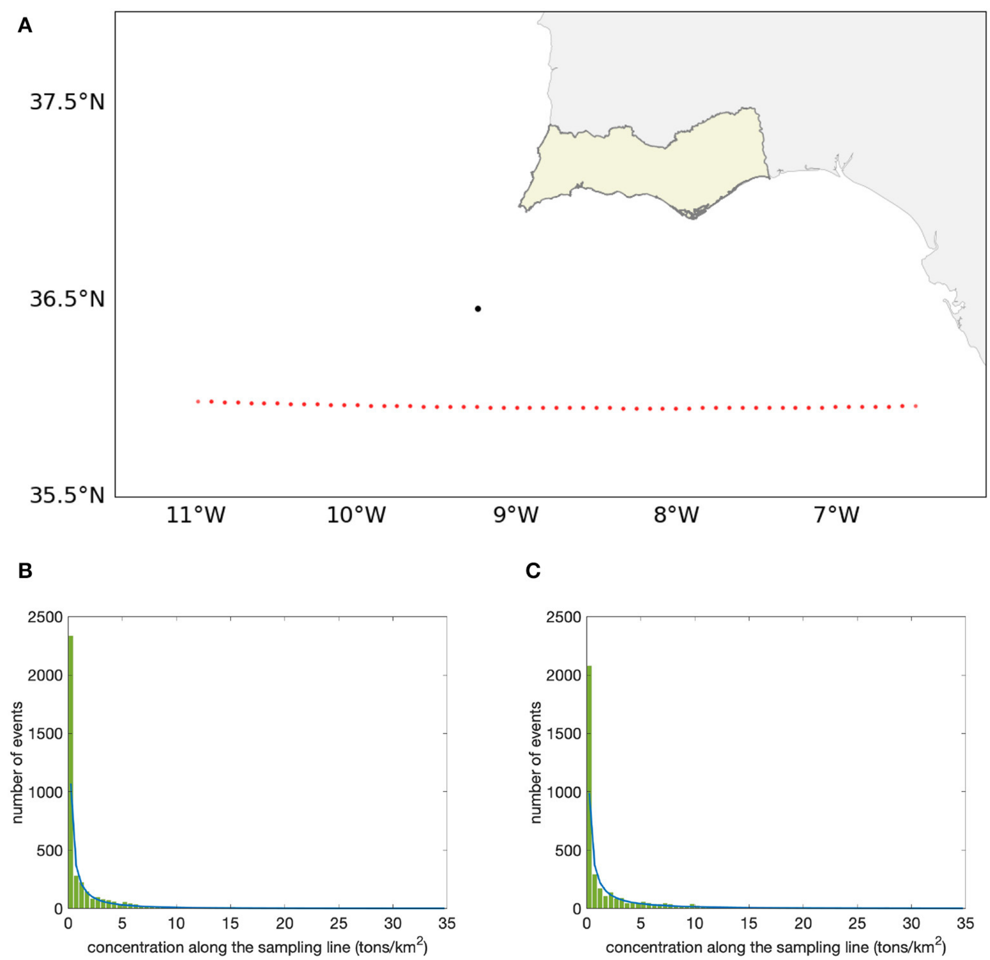

FIGURE 2 | (A) Map of the single release point (black dot) off the Algarve with the "sampling line" drawn in red dots. Histograms and Weibull function estimation for oil concentrations obtained at the "sampling line" **(B)** for the weathering experiment, which includes transport and weathering processes and **(C)** for the no-weathering experiment.

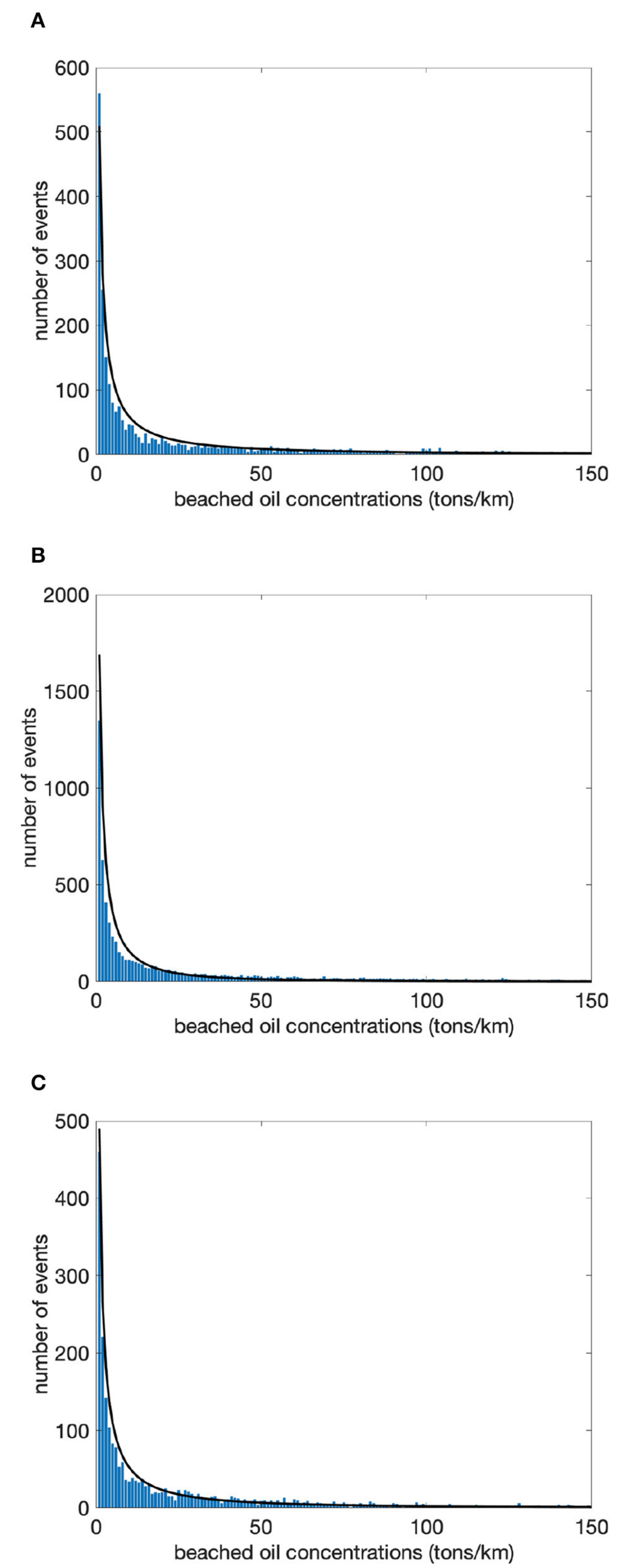

FIGURE 3 | Histogram (blue bars) and Weibull distribution function estimation (solid black line) of beached oil concentrations from the ensemble simulations for **(A)** the Eastern Atlantic Archipelago, **(B)** the Western Atlantic Island, and **(C)** coast of Bahia (Brazil).

TABLE 3 | Weibull distribution parameters emerging from the maximum likelihood estimation.

Weibull parameters	Eastern Atlantic Archipelago	Western Atlantic Island	Bahia (Brazil)
Scale (η)	5.1 ± 0.6 tons/km	5.86 ± 0.75 tons/km	4.2 ± 0.5 tons/km
Shape (β)	0.362 ± 0.008	0.377 ± 0.009	0.377 ± 0.008
Mean (μ)	23 tons/km	23 tons/km	17 tons/km
Standard deviation (σ)	85 tons/km	80 tons/km	58 tons/km

The error in the scale and shape parameter has been computed by a bootstrap algorithm subtracting $\sim 10-20\%$ *of the input data. In addition to the scale and shape parameters, we calculate also the mean and standard deviation of the estimated distribution, defined as:* $\mu = \eta\,\Gamma\left(1+\frac{1}{\beta}\right)$ *and* $\sigma^2 = \eta^2\left[\Gamma\left(1+\frac{2}{\beta}\right)-\Gamma\left(1+\frac{1}{\beta}\right)^2\right]$, *where* Γ *is the Gamma function.*

seems likely that an active ocean tracer, such as oil, would fit a statistical distribution with a heavy tail, i.e., have a distinct non-Gaussian behavior.

The estimate of (2) to the sampling line events is shown in **Figure 2** demonstrating that Weibull is a good distribution function for the oil arriving at the sampling line. The estimated shape parameter value is $\beta = 0.399 \pm 0.005$, indicating a large peak at the low concentrations, for the weathering and no-weathering simulations. The scale parameter is instead different between weathering and no-weathering simulations because it is connected to the distribution mean value and it is $\eta = 0.50 \pm 0.03$ tons/km^2 and $\eta = 0.93 \pm 0.06$ tons/km^2 for the weathering and no-weathering experiment respectively.

We conclude that the oil concentration distribution from a single point source release and along an offshore sampling line has the form of a Weibull distribution that is peaked around the lowest concentrations.

5. THE BEACHED OIL CONCENTRATION PDF

In this section we will show that, notwithstanding the complexity of beached oil processes, the beached oil distribution along the coastlines of three different areas in the Atlantic has a universal structure similar to the surface oil PDF found along the sampling line in the open ocean (section 4).

Following the experimental set up described in section 3, we collect the beached oil parcels on each segment of the coastline for each run. The number of RP points in the three areas are reported in **Table 2** together with the number of simulations and beached oil events in each area. In **Figure 3**, we present the histograms of the beached oil concentration distributions for the Eastern Atlantic Archipelago, Western Atlantic Island, and state of Bahia (Brazil) coastlines.

The Weibull distribution function seems to estimate well the beached oil data, even if some discrepancies appear at concentration values between 0 and 50 tons/km. The scale, η, and shape, β, parameters are reported in **Table 3**. It is clear again that the shape parameter is less than 0.5 for all the three areas, showing

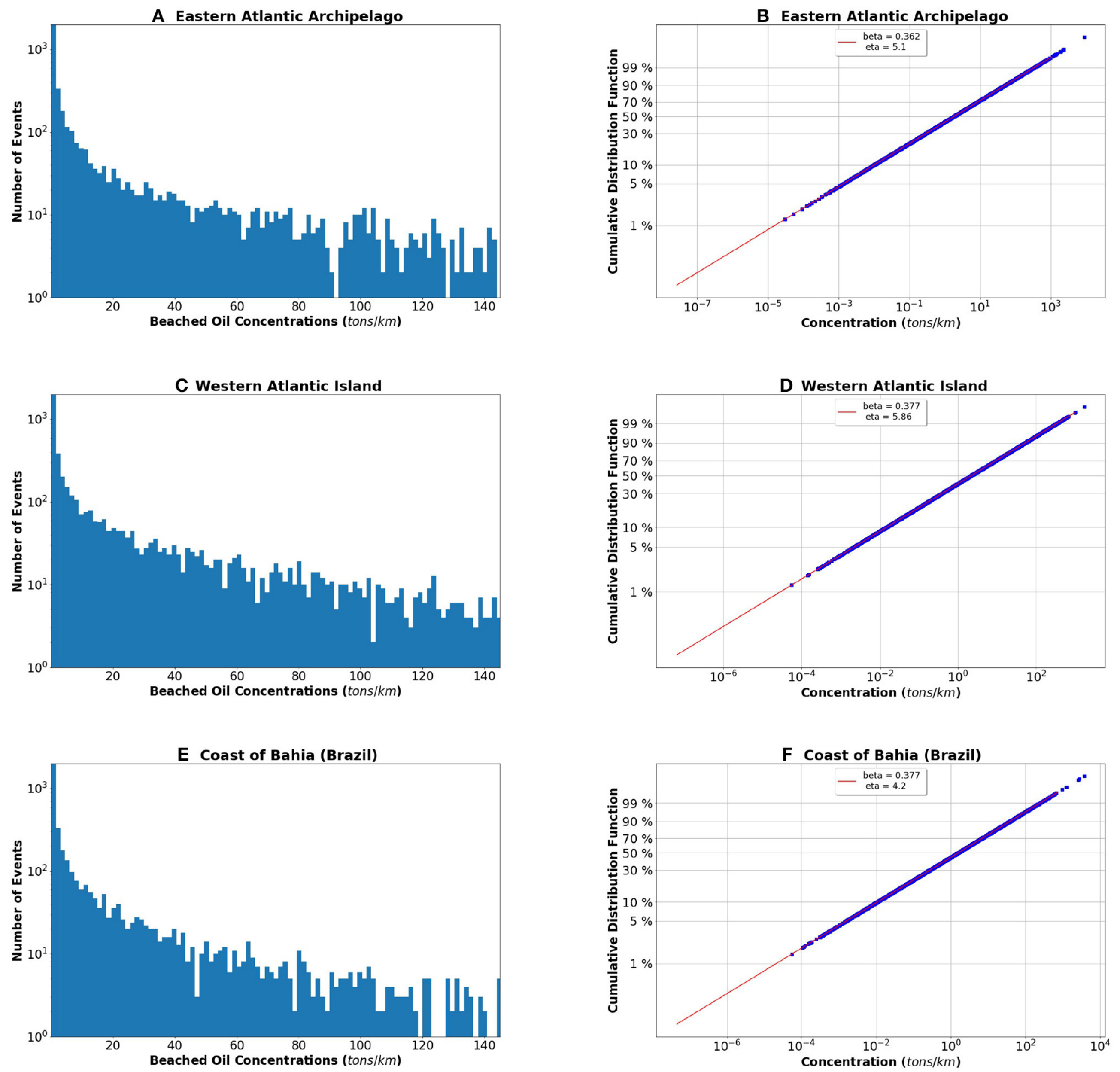

FIGURE 4 | (A,C,E) The histogram for each of the three study areas, the Eastern Atlantic Archipelago, the Western Atlantic Island, and the Bahia coasts respectively, displayed with a logarithmic scale. **(B,D,F)** The Weibull plot with the cumulative distribution function $F(x) = 1 - e^{-(\frac{x}{\eta})^{\beta}}$ in the ordinates and the concentration x in the abscissa, both displayed on a logarithmic scale.

that the maximum number of events occur at the lowest values of the distribution. It is puzzling that the shape value is the same for all the areas, within the calculated errors, and it is also similar to the one found for the open sea conditions in section 4. We argue that our results show that the beached oil distribution does not depend on the specific coastal geometry but only on the type of flow field conditions that impinge on the coasts. The coastal currents are themselves affected by the type of bathymetry and coastline, so the dependence is indirect. Furthermore the beached oil PDF depends also on the winds, both directly and indirectly: the former through the transformation processes and the latter because currents are themselves driven by winds.

The histograms shown in **Figure 3** could be also displayed on a logarithmic scale that would enhance the fat tail structure of the three distributions (**Figures 4A,C,E**). To check visually if our data set comes from a population that would logically be fit by a 2-parameter Weibull distribution we use the Weibull plot (Nelson, 1982). It is clear from **Figures 4B,D,F** that the assumption of a Weibull distribution is reasonable and that there are no outliers.

In **Table 3**, we list also the mean and standard deviation of the distributions from the estimated Weibull function. The Eastern Atlantic Archipelago and Western Atlantic island have similar "mean" value of beached oil concentration, i.e., 23 tons/km, while Bahia $\sim$ 30% less, i.e., 17 tons/km. We conclude that the mean value of beached oil, due to surface current transport and wind induced transformations, is quite different between the two northern Atlantic areas and Bahia.

The mean of Weibull distributions with a β parameter smaller than 1 is shifted toward low concentrations. Thus, the distribution mean is not really sufficient to assess the hazard which should consider also the large events in the PDF tail. In order to do so we need to consider the cumulative distribution and define an appropriate indicator.

6. A GENERAL INDICATOR OF COASTAL OIL SPILL HAZARD

We will now show how beached oil concentration PDFs can be used to describe the hazard in a general way, paving the way for a quantitative method to answer a key question related to the management of the oil spill risk: what is the beached oil spill hazard for different areas and which area has the highest hazard?

Having now the PDF which describes the beached oil events from a set of RPs, simulating potential releases from accidents, we can define an index that contains the information about the large pollution events, concentrating on the tail of the Weibull distribution. It is indeed reasonable to assume that the hazard of large oil releases near the coasts should be evaluated by estimating the extreme events of the distribution.

To describe the extreme events it is then necessary to find how many events are contained in the tail with respect to the overall distribution events. To this end, we can use the Weibull tail distribution H, defined as:

$$H = 1 - F(x_{cut}) = e^{-(\frac{x_{cut}}{\eta})^{\beta}} \tag{3}$$

where $F(x_{cut})$ is the Weibull cumulative distribution function for the beached oil concentration x for which $x \leqslant x_{cut}$. When H is large, it means the distribution tails contain a relatively large number of high concentration events, thus the hazard is large while the contrary is true for small values of H.

The key parameter here is the threshold value x_{cut} chosen for the estimation of extreme events. For illustrative purposes, a concentration of 25 tons/km was applied following the International Tanker Owners Pollution Federation (ITOPF) Technical Information Paper on *"Recognition of oil on shorelines"* (ITOPF, 2011). The threshold is proposed for classifying impacted coastlines as heavily oiled.

The tail distribution H was computed for each area (**Table 4**) for the chosen value of the threshold. The Western Atlantic island emerges as an area of relatively high hazard with respect to the Eastern Atlantic Archipelago and the Bahia areas. This means that large beached oil events are more likely to occur in the Western Atlantic island, about $\sim$ $12-14\%$ more frequently than in the Eastern Atlantic Archipelago and the Bahia coasts, within the limitations of our ensemble simulations, in particular the length of the simulation and the distribution of RP.

TABLE 4 | Beached oil hazard index calculated with Equation (3) for the three areas and the threshold value of 25 tons/km.

Area	Beached oil Hazard index
East Atlantic Archipelago	0.16 ± 0.01
Western Atlantic island	0.18 ± 0.01
Bahia (Brazil)	0.14 ± 0.01

7. CONCLUSIONS

This paper develops a straightforward and objective method to quantify the coastal oil spill hazard based on ensemble oil spill experiments which sample the uncertainties associated with oil spill accidental releases in the ocean areas next to the coasts. Ocean current variability at the mesoscales affects the flow field and in order to have a realistic sampling of the oil transport variability, ensemble simulations have been extended for an entire year, sampling the seasonal variability of the flow field at mesoscale permitting resolution.

It has been found that both oil in the open ocean and beached oil concentrations are successfully described by the Weibull distribution. Such distribution is characterized for these cases by a shape parameter which is less than 1, thus having the maximum beached oil events at small concentrations. The large beach oil concentrations are contained in a "fat tail" which characterizes this distribution. Concentrations at the coasts span two orders of magnitude values, from 1 to 100 tons/km. Previous studies have indicated that also the winds, currents, and tracers in the atmosphere and oceans distribute as Weibull but this has never been shown for active ocean pollutants, such as oil.

Furthermore, the paper proposes a new hazard index for beached oil which allows to intercompare different world ocean areas and estimate the different hazards. The index is based upon the fit of a Weibull PDF to the simulation data and extracting the tail distribution for large events.

We would like to point out that the present study has potential limitations due to the relatively low number of uncertainties considered in our simulations. In the future, experiments should probably vary the simulation length, should check different oil types and beaching algorithms and use different meteo-oceanographic conditions including higher frequency winds and tides.

This paper does not discuss how exactly the flow field modulates the oil spill hazard through advection-diffusion processes in the ocean. Nor our oil spill hazard estimates for the three regions examined are conclusive for future management decisions, especially due to the limited number of oceanographic-weather conditions considered in the study. Such estimates can be obtained with the methodology described here, but will require more extensive investigation and analyses. It is important here that a common quantitative framework for intercomparison of different world ocean areas has been found. Furthermore, to evaluate risk, vulnerability data need to be acquired and used to modulate the hazard that we have computed here (Alves et al., 2016a).

Future work will consider an in-depth study of the ocean flow field parameters and how they modulate the coastal oil spill hazard. Additionally, the spatial resolution of the oceanographic-weather fields used in our experiment offers reliable answers only for mesoscale dominated flows and coarsely resolved coastal morphology and bathymetry. The increase in resolution of the ocean and atmospheric models, and therefore their ability to reproduce smaller scale features and higher frequency variability, should impact the oil concentration PDF and, consequently, the oil spill hazard estimates. Furthermore, uncertainties connected to the amount of oil released should be also considered in the future, i.e., cases of larger oil spill releases.

DATA AVAILABILITY STATEMENT

The datasets generated for this study are available on request to the corresponding author.

AUTHOR CONTRIBUTIONS

All authors contributed to conception and design of the study. AS generated the database and wrote the first draft of the manuscript. NP and AN wrote sections of the manuscript and fully revised the manuscript. FT made the additional figure on the revised manuscript and discussed the meaning. All authors contributed to manuscript revision, read and approved the submitted version.

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Linking User-Perception Diversity on Ecosystems Services to the Inception of Coastal Governance Regime Transformation

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In this paper we explore the challenges for transforming a wide and fragmented coastal governance system toward an ecosystem-based regime by translating shared values of nature into radically novel territorial development policies at highly disputed seascapes. We report an official coastal management institutional experiment in South Brazil, where direct ecosystem users (fishers, miners, mariculture, tourism and leisure, and aquatic transport agents and researchers) perception and classification of ecosystem services (ES) was assessed during 19 collaborative sectoral workshops held with 178 participants from six coastal cities surrounding Babitonga Bay estuarine and coastal ecosystems (Santa Catarina state, South Brazil). Participants collectively enlisted the benefits, rights and resources (or services) they obtain from these ecosystems, rendering a total of 285 citations coded to conventional ES scientific typologies (127 ES grouped in 5 types and 31 subtypes). We explore patterns in ES classificatory profiles, highlighting ecosystem user's salient identities and exploring how they shape political actions in relation to the implementation of an ecosystem-based management regime. Food (provisioning service), tourism/leisure, employment, work and income (cultural services) as well as transportation (e.g. vessels, ports and navigation) (cultural/people's services) are perceived by all user groups, and hence consist the core set of perceived shared values amongst direct ecosystem users to inform future transformation narratives. Differences in perception of values amongst user groups combined with high levels of power asymmetry and fragmentation in decision-making, are steering the analyzed system toward an unsustainable pathway. The governance regime has been largely favoring subsets of services and unfair distribution of benefits, disregarding a more diverse array of real economic interests, and potential ecological knowledge contributions. Our integrative and deliberative ES valuation approach advances understanding of critical features of the scoping phase of ES assessment initiatives in coastal zones. We provide empirically grounded and theoretically informed suggestions for the promotion of local knowledge integration through combination of methods that supports transformational research agendas. This paper establishes new groundwork to fulfilling alternative visions for the regional social-ecological system transformation to a more socially and ecologically coherent and equitable development trajectory.

Keywords: perception, ecosystem-based management, shared values, social-ecological system, stakeholders, Brazil

INTRODUCTION

Ecosystem Services Assessments on the Crossroads

Ecosystem services (ES) are commonly defined as benefits obtained from the environment by humans and are critical to human survival, livelihoods, well-being, and quality of life (Millennium Ecosystem Assessment [MEA], 2005). Understanding and integrating the diversity of human perceptions and agency on coastal and marine seascapes and related ES into governance processes remains a critical challenge to avoid escalating conflicts over marine resources in the Anthropocene (Folke, 2006; Liquete et al., 2013; Aswani et al., 2017). Our society lives a dilemma. While we depend on coastal-marine ES and states actively promote the ocean as the new global economic development and growth frontier (Bennett et al., 2019), anthropogenic factors have already affected their resilience and, therefore, are increasingly compromising sustained availability of these services at regional levels (Gattuso et al., 2018).

Coastal social-ecological systems (SES) are interface regions, rendering them higher complexity to govern a variety of dynamic, highly uncertain socioeconomic, political, and biophysical interactions and flows (Zaucha et al., 2016). These features, and the high levels of historical path dependency and self-identification in land-sea territories, most often hinder the much needed, rapid transformations in their prevailing development paradigms (Zaucha et al., 2016).

The complexities of coastal-marine systems thus require regarding them as coupled SES, an interdisciplinary approach that regards separations between the social and natural systems as artificial and arbitrary (Berkes and Folke, 1998). Thereby, understanding how human perception-driven standpoints relate to ES is an important part of understanding SES dynamics and complexity, i.e. since preference of certain services may affect their availability and the very structure of ecosystems into the future. This requires acknowledging humans and human agency as an integral, embedded part of ecosystems and therefore highlighting their perception, interaction, joy, and interference capacities, as natural ecosystem processes: a *humans-in-ecosystems* perspective (Davidson-Hunt and Berkes, 2003). This approach considers humans as both co-producers and consumers of ES (Raymond et al., 2017) that, in turn, result from the combination or interaction of natural (including human, social, and built) capitals (Costanza et al., 2017).

Since the worldwide boom in ES conceptual research and application following the Millennium Ecosystem Assessment in 2005, the link between ES and environmental governance has been widely discussed (Abson et al., 2014; McDonough et al., 2017). Ever since, worldwide application and development of the ES toolboxes by several organizations, for initiatives aimed primarily at conducting services valuation assessments have increased tremendously. But challenges in the science and application of ESs remain, such as conflicting terminology, classification schemes, research methods and reporting requirements (McDonough et al., 2017). It is within this diversity of understanding and application realm that scientists have continuously pursued development of alternative frameworks, with the ultimate intent of improving and adjusting ES concepts and typologies for practical application (Costanza et al., 2017; Díaz et al., 2018).

Facing the Practical Challenges of Integrated and Deliberative Valuation Approaches

Our paper combines integrative (of diverse values) and deliberative (participatory reasoning and awareness-building) elements in research-design, to generate collective understanding about shared values of nature and build practical knowledge for sustainability in a highly disputed seascape. This is in accordance with strong, recent calls by the International Panel for Biodiversity and Ecosystem Services (IPBES) for the evolution of frameworks that are better able to accommodate alternative worldviews and bridge scientific with local/indigenous ecological knowledge systems (Díaz et al., 2015).

Costanza et al. (2017) argue that ecosystem users should ideally collaborate in ES modeling and scenario planning through transdisciplinary teams and strategies, in order to assure relevancy of application in real policy contexts at multiple time and space scales. Consistency will partly evolve from further understanding the underlying determinants of how a “shared value” is socially constructed and represented in ES assessments and policy arenas (Vatn, 2009).

Valuation is not a last nor optional step in ES assessments, but span over multiple steps – from the choice of value types and of terminology, selection of social actors to engage with, methodological decisions (tools and measurement units), and choice of which ES are to be included in research (Martín-López et al., 2013; Jacobs et al., 2016; Boeraeve et al., 2018). Further attention should also be placed on participatory methods capable of recording less tangible cultural ES and non-material values (Raymond et al., 2009; Milcu et al., 2013; Fish et al., 2016; Boeraeve et al., 2018), and including them alongside other services in governance processes that embeds the diversity of perceptions in transformations toward sustainability (Chan et al., 2012; Larson et al., 2013; Jacobs et al., 2016). The driving rationale is that integrating peoples' values and perceptions into planning may allow for the build-up of more effective and compatible science-policy exchange, by matching the multiplicity of uses by different actors with the maintenance of ES through more equitable processes and outcomes (Larson et al., 2013).

Nonetheless, few studies characterize how the ES concept articulates with local ecological knowledge systems (Oliveira and Berkes, 2014). Perception can be defined as an experiential process where organisms (in this case humans) see, test and feel the components of a lived moment (Whyte, 1977); or the process of translation and reconstruction of brain stimuli and signals captured and encoded by sensations (Morin, 2000). Some of the earliest ES models already acknowledged how just a small percentage of ES are usually perceived and therefore valued by humans (e.g. Costanza and Folke, 1997). We now know that the diversity and structure of patterns in human perception of nature can vary according to the types of ecosystems analyzed (Costanza, 2000; Casado-Arzuaga et al., 2013); age and education of people involved (Blayac et al., 2014); social position and occupation (Oliveira and Berkes, 2014); and all factors affecting methodological options underpinning ES research (McNally et al., 2016; Simpson et al., 2016).

Jacobs et al. (2016) makes a strong case for integrative valuation approaches and actually proposes a new valuation school aimed at integrating diverse values of nature in resource and land use decisions. They outline the key challenges that need to be overcome by this emerging science-policy field, which we summarize in the following eight challenges: developing a strong interdisciplinary basis (1); combination (2); application of appropriate methods (3); ethical consideration about the impact of research for embedded sociopolitical (4); governance realities (5); the challenge of communicating complexity and uncertainty about values of nature to stakeholders and decision-makers (6); issues of equity and power asymmetries (certain values benefit actors with more power) (7); and the higher costs and breadth of time- and data-consuming nature of such research processes (which might be seen as less efficient) (8). Studies seeking to face such challenges are under development in several places, but they most often do not address all the challenges at once (Jacobs et al., 2016). While challenges 4 and 5 are given structural properties of SES and as such modifying them are perhaps to be regarded as long-term research-policy outcomes; all others stand as options that can be embedded in inter- and transdisciplinary research design early on their inception in real SES. Our paper reports a highly interdisciplinary, on-going research-action project attempting to consider all such project design challenges to face real structural transformations in sociopolitical and governance features of a coastal-estuarine SES in the long-run.

Transforming Coastal-Marine Social-Ecological Systems

The accelerating crisis in common pool environmental resources worldwide has impelled recent scholarship to understand and inspire the achievement of lasting change in the way SES are organized (Gunderson and Holling, 2002; Folke et al., 2010; Patterson et al., 2017). More now than ever in human history, transformative change is urgently needed in how people and institutions interact with coastal systems (Glaser et al., 2012). In the context of our research, we highlight the pressing challenge for rapid shifts in how coastal and marine governance evolves, toward regimes that can deliver more socially and ecologically coherent outcomes (Young, 2010; Westley et al., 2011, 2013). The inception (step-zero) of radically novel area-based interventions is one of the most critical challenges of any given coastal-marine SES trajectory (Chuenpagdee et al., 2013).

For instance, most countries have developed national marine protected areas (MPAs) frameworks to promote a range of area-based marine management objectives including spatially and temporally sustainable resource management. Given that only about 3% of all oceans are governed by MPAs, a real big challenge for marine conservation goes beyond improving effectiveness of existing MPA systems; but also to create new ones and broadly increase capacities to govern coastal-marine systems beyond MPAs through "other effective area based conservation measures" (OECMs) (Laffoley et al., 2017). OECMs are defined as: "*a geographically defined area other than a Protected Area, which is governed and managed in ways that achieve positive and sustained long-term outcomes for the in situ conservation of biodiversity, with associated ecosystem functions and services and, where applicable, cultural, spiritual, socioeconomic, and other locally relevant values*"(CBD Recommendation No 22/5, July 2018). The implementation of OECMs resonates with recent calls for the planning of networks of MPAs to be consciously promoted as "policy experiments" (Fox et al., 2013) by research-action projects, through continual models of stakeholder engagement and learning (Reid et al., 2016) that includes coastal-marine areas within and between formally designated MPAs.

In face of the above challenges in ES-based research and policy – this paper analyses the Babitonga Bay estuarine SES (South Brazil) study case, one that has been undergoing rapid transformation in the way it is governed and therefore has been endorsed by the Brazilian state as "policy experiment" – to our knowledge the first pilot marine OECMs in the country. We will explore how diverse patterns in perception of values of nature by direct ecosystem users, affects the inception of new, territorially bonded "shared values" discourse as a key feature for the transformation of the currently fragmented toward an ecosystem-based coastal governance regime. Our paper will highlight the lessons learned in relation to the scoping phase of coastal-marine ES assessments and, more broadly, the potential contribution of integrative and deliberative ES valuation approaches to coastal-marine ecosystem-based policy-making.

MATERIALS AND METHODS

Driving Social-Ecological Transformations in Babitonga Bay

Babitonga Bay is on the northern coast of the state of Santa Catarina (Brazil). It is surrounded by six coastal municipalities (**Figure 1**) and includes the largest metropolitan region of the state, around the city of Joinville (about one million inhabitants).

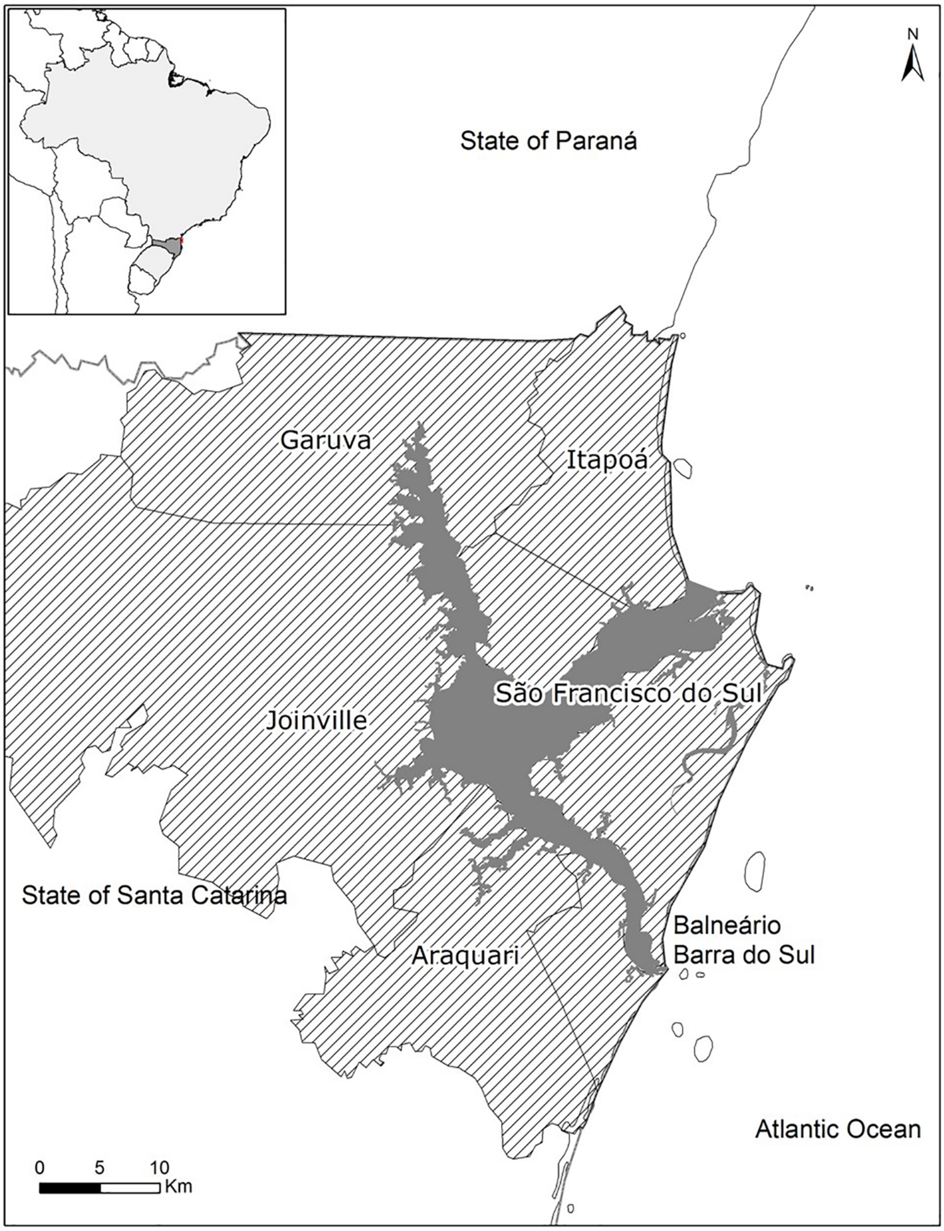

FIGURE 1 | Babitonga Bay and its six surrounding municipalities (North of Santa Catarina – Southern Brazil).

The estuarine system has an area 1400 km^2, and the largest mangrove area in southern Brazil, with 130 km^2 (Barros et al., 2008), or 75% of the state mangrove cover (MMA, 2002). This estuary connects to the ocean through one channel with an extension of 1.7 km, and also comprises sandy beaches, 83 islands, stone slabs, and sand banks (Instituto Brasileiro de Meio Ambiente e Recursos Naturais Renováveis [IBAMA], 1998).

The ecological functions of Babitonga Bay allow the survival of several species, temporary (migrant) or resident, including 28 endangered or particularly valued commercial fishes (Gerhardinger et al., 2006; Gerhardinger et al., in press) and the critically endangered porpoise (*Pontoporia blainvillei*; Cremer and Simões-Lopes, 2005). The Bay houses diverse activities, such as agriculture, tourism and leisure, mariculture, fisheries, and port and industrial activities (Barros et al., 2008). Due to the urbanization, port activities, and the discharge of untreated sewage, some areas are highly polluted and contaminated by fecal sterols (Martins et al., 2014) and organic matter (Barros et al., 2010). Both inner and outer-bay coastal seascapes are used by over 1,700 fishers from the six surrounding municipalities. Other direct users are related to two ports, two sand mining companies, mariculture (aquaculture parks), and tourism and leisure operators, including marinas. The sharing of the area by different users generates pressures and conflicts on the ecosystem. The power asymmetry and the fragility of over a handful of ongoing environmental licensing processes of large coastal infrastructure (e.g. new ports) offers a "*...perfect atmosphere for political speculation and unethical bargaining* [of territorial rights] *...and proliferation of fallacious information...*", also reflecting the lack of integration of local actors' perceptions toward a more equitable development scenario (Gerhardinger et al., 2018a).

Since 2015, collaborative activities have been developed in coastal cities around Babitonga Bay through a growing network of over 60 organizations involved in socio-environmental projects, mobilizing direct and indirect resource users, governmental and NGOs into a novel coastal governance architecture for the area (Gerhardinger et al., 2018b). Gerhardinger et al. (2018b) have recently analyzed the Babitonga Bay SES trajectory, suggesting that recent interventions have put the SES on the move toward transformation, i.e. tipping the SES to a "hazy-to-transparent" phase of the SES following Westley's et al. (2013) theory of transformative agency (TAT). Even though a comprehensive toolbox for integrated coastal management policies were already available to local decision-makers, before the project started, the SES was suffering with the ruling of a largely fragmented and sectoral governing approach reported above.

Three years later, a humans-in ecosystem-based vision for Babitonga Bay area-based governance is now being pursued by members of a newly established, autonomous multi-stakeholder forum named Pro-Babitonga Group (PBG). This forum is formed by representatives of public and societal sectors and have been endorsed by Brazil's Federal Action Plan for the Coastal Zone as a regional integrated coastal management policy experiment. Gerhardinger et al. (2018b) suggests the very existence and operation of PBG indicates that old ways of governing are losing dominance, and institutions and beliefs are opening to reinterpretation in a novel system which enables the exchange of ideas, evaluation of scenarios and definition of new ecosystem-based governance trajectories. This very special policy condition offers a rare opportunity to translate the diversity of resource user perceptions on ES in the crafting of a new, more socially and ecologically equitable and coherent vision for the future of the SES.

Selection of Participants

Research co-design started in June 2015 with a workshop with researchers, representatives of national and municipal public agencies (Instituto Chico Mendes de Conservação da Biodiversidade – ICMBio, Instituto Brasileiro do Meio Ambiente e Recursos Naturais – IBAMA, local governments) and socio-environmental organizations. Through this workshop, engagement with five groups of direct ecosystem users were deliberately prioritized: artisanal fishers, mariculture agents (oyster and mussel cultivation), aquatic transport agents (representatives from the port, collective maritime transportation companies, barge, and petroleum transportation companies), miners and, tourism and leisure agents (marinas, passenger boats, owners of sports fishing boats).

The strategies for selection of workshop participants sought to guarantee representativeness of groups and varied according to number of people/institutions in each group, in each of the six municipalities surrounding Babitonga Bay (see **Supplementary Appendix S1**, with the detailed description of group selection and mobilization).

Data Collection

This paper reports the results from the first round of an ongoing ecosystem-based marine spatial planning workshop series, a process driven by non-state actors during the early implementation-phase of a continual and long-term multi-actor engagement model (Reid et al., 2016). Participatory data-collection workshops were designed and replicated with all five direct Babitonga Bay ecosystem users and researchers in separate sessions, in each municipality, after prior informed consent of the participants. All the workshops followed the same methodology with a minimum of two facilitators.

In order to elicit ES types and to understand how the groups perceive ES from the socioecological system, we used the inductive word "Benefit" (**Figure 2**) – referring to the product that nature provides for humans, and because some researchers consider it to be synonymous of ES (e.g. Millennium Ecosystem Assessment [MEA], 2005). During preliminary assessments, local fishers' responses to "Benefit" enacted their perception of *governmental benefits* (e.g. insurances, retirement). Therefore, we used the complimentary inductions "Access Rights" and "Resource" (in respective order) to expand the identification of ES. Thus, participants were invited to argue about the benefits they obtain from nature where they live, what are their access rights and what resources they use. The first mention of every citation was recorded on notecards and organized in a panel board below each inductive word heading.

Data Analysis

Our analysis sought to contrast local classificatory systems (emic: the perspective of investigated social groups/informants) with scientific knowledge (etic: perspective of researchers) (Posey, 1987), thus transforming and encoding popular knowledge about the environment based on scientific theories, into

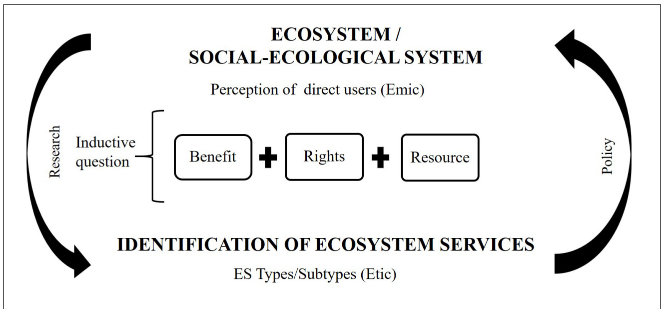

FIGURE 2 | Integrative/deliberative data-collection process (n = 19 workshops; 178 participants) conducted to understand the ecosystem services (ES) perceived by the direct users and to initiate the construction of new policies for the Babitonga Bay ecosystem (Brazil).

ongoing decision-making processes. Therefore, we contrast local knowledge with the Millennium Ecosystem Assessment concept of ES as "*benefits obtained from the environment by humans*" (Millennium Ecosystem Assessment [MEA], 2005); and the four basic types of ES (provisioning, regulating, supporting, and cultural). All citations recorded during participatory workshops were systematized, categorized and counted as responses to benefit, access right or resource. We standardized citations, coding them into groups of similar meanings. For example, *bathing* and *swimming* were considered swimming; *employment* and *work* as employment; *fun* and *outings* as leisure; *forest* and *bush* as vegetation. During the coding process, we acknowledged that the MEA's framework did not fully accommodate the diversity of human-environment relationships (see also Wallace, 2007; Oliveira and Berkes, 2014). Kenter et al. (2016) notes that straight classification of cultural ES as benefits is often problematic (i.e. they can be intangible, experiential, and identity-based or idiosyncratic), raising particular axiological and ontological issues that calls for deliberative and non-monetary valuation approaches. Therefore, we adapted Raymond et al. (2009) refinement of the Posey (1987) typology; hence, when accessing emic perceptions, we used a "people's" services subtype within Cultural ESs that enabled the full consideration of the local ecological knowledge of the users, about the services they report from the ecosystem. People's ES are considered here as cultural benefits derived from human agency. They refer to values and threats to the ecosystem, as informed by workshop attendants, but not straightforwardly falling in the conventional ES Cultural category. Thus, our dataset was coded in the following types of ES: provisioning, regulating, supporting, cultural, and cultural/people's as a special type of cultural ES (**Table 1**).

TABLE 1 | Definitions of types of ecosystem services used in this article, adapted from Raymond et al. (2009) and Costanza et al. (2017).

Service Type	Definition
Supporting	The very structure that supports life and all other services, they are basic ecosystem processes such as soil formation, primary productivity, biogeochemistry, nutrient cycling and provisioning of habitat
Regulating	Derives from the combination of natural with built, human, and social capital to produce flood control, storm protection, water regulation, human disease regulation, water purification, air quality maintenance, pollination, pest control, and climate control
Provisioning	Derives from the combination of natural with built, human, and social capital to produce and extract food, timber, fiber, or other "provisioning" benefits
Cultural	Derives from the combination of natural capital with built, human, and social capital to produce recreation (e.g. beach, swimming, boat touring), esthetic (scenic beauty, landscape), knowledge (information and education), cultural identity (e.g. fishing, diversity of local traditions), sense of place (e.g. satisfaction and pleasure to live in a given place), legacy (e.g. taking what one needs for sustenance and survival, services for future generations) or other "cultural" benefits
Cultural/People's	Human beings are regarded as agents that transforms and generates benefits in the ecosystem (including natural and social properties). Therefore, we use this category to embrace cultural benefits directly derived from human agency in social-ecological system and constructions in nature: physical structures enabling direct access to services (e.g. logistics, boats, ports, industries, roads, shipyards), sharing an economic (e.g. job creation, income generation, profiting) and social organization purpose (e.g. institutions, laws such as closed fishing season and retirement, political dynamics, supervision)

RESULTS

The 19 workshops with direct Babitonga Ecosystem users and researchers mobilized 178 participants (see **Supplementary Appendix S1**). We obtained a total of 285 ES citations (average of 15 citations per workshop), 210 were in response to the word Benefit (Average = 11/workshop), 57 in response to Access Rights (Average = 3/workshop), and 18 elicited by the word Resource (Average = 0.95/workshop).

The use of three complementary inductions therefore contributed to increase the overall number of citations – even though we excluded repetitions leading to gradual exhaustion of new valid citations. Researchers were outstandingly above average in total number of citations in a single workshop ($n = 37$).

The citations were coded into 127 distinct ESs, the richest being: leisure ($n = 13$), tourism ($n = 12$), fish ($n = 11$), water ($n = 9$), fisheries ($n = 9$), navigation ($n = 8$), crabs ($n = 7$), and survival, food, air, oyster and navigability ($n = 5$ each). We obtained 45 (16%) citations of fish or crustacean species, representing at least 16 different species.

We identified a total of 31 ES subtypes, including: Regulating = 3; Supporting = 3; Provisioning = 5; Cultural = 20; Cultural/People's = 9 (**Table 2**). During the ES type and subtype assignment process, we took several steps to harmonize classifications with overlapping meaning and avoid typological misrepresentations in further analysis. Therefore, ten citations were disregarded because they were similar to others mentioned under different inductive stimuli. We removed citations such as "quality of life" ($n = 7$), "well-being" ($n = 1$) and "health" ($n = 2$) in response to inductions with the word "benefit" ($n = 8$) and "access rights" ($n = 2$), because they resulted from the combination of subsets of benefits pertaining to all categories. Citations could be assigned to two types of ES, for example, mariculture and agriculture were classified as a provisioning in the food subtype and in "*People*" as a source of income, for producing food from man-made production and cultivation structures rather than simply extracting what is produced in nature.

We obtained a total of 317 classifications (the 270 citations plus 52 citations that were assigned to more than one subtypes). Among the 31 subtypes, eight presented only one citation (**Table 2**).

Cultural and cultural/people (62% of all classifications) and provisioning (29%) were the most cited types of ES overall. The former was the most frequent type to all but fishers who cited more provisioning ESs (**Figure 3**). Regulating and supporting services accounted for the lowest numbers of classifications. They were seldom referred by direct users other than by researchers,

TABLE 2 | Structure of Ecosystem Services classification profiles by direct resource users (N = number of workshops) of Babitonga Bay (Santa Catarina, Brazil).

Ecosystem service type	Ecosystem service subtype	Researchers *N* = 1	Fishers *N* = 9	Mariculture *N* = 1	Tourism and leisure *N* = 5	Mining *N* = 2	Aquatic transport *N* = 1
Supporting	1. Maintenance of life cycle	3	0.7	1	0.2	0.5	
	2. Maintenance of genetic diversity	1	0.1		0.6		
	3. Nutrient cycling	1					
Regulating	4. Air quality	1	0.2		0.2	0.5	
	5. Climate regulation	1					
	6. Regulation of erosion					0.5	
Provisioning	7. Food	3	5	2	1	1	1
	8. Genetic resources	2	5	1	0.6		
	9. Water	1	0.1	1	0.4	0.5	
	10. Mineral resources	1	0.2			1.5	
	11. Geomorphologic resources						1
Cultural	12. Leisure and tourism	3	0.9	3	2.4	3.5	2
	13. Cultural and historical patrimony	3	0.7	1	0.4	1	
	14. Legacy and existence		0.4	2	1.2	0.5	
	15. Aesthetic, inspiration and contemplation	2	0.7		0.6	1	
	16. Sense of place	1	0.6		0.2	0.5	
	17. Education and knowledge system				0.2		1
	18. Livelihood		0.1		0.4		1
	19. Social relations	1			0.2		1
	20. Communication and information	1	0.1		0.2		
	21. Hunting					0.5	
	22. Spirituality		0,1				
Cultural/People's	23. Economic viability	2	5.2	3	0.6	1	1
	24. Infrastructure and logistics	3	0.8	2	2.6	1.5	2
	25. Assistentialism		0.7				
	26. Planning		0.2		0.8		
	27. Strategic geographic position				0.2		2
	28. Supporting institution and legislation		0.1			1.5	
	29. Financing		0.2				
	30. Opportunity		0.1				
	31. Politics				0.2		
Grayscale of average number of citations per workshop:		[0 - 0,5]	[0,5 - 1]	[1 - 1,5]	[1,5 - 2,5]	[2,5 - 3,5]	>3,5

who mentioned several of such types as important ESs. Aquatic transport agents did not refer to any regulating and supporting ES, while mariculture agents did not mention regulating services.

We adapted the *framework* from Raymond et al. (2009) including a gradient of ES. On the left side (**Figure 4**), we show ESs predominantly deriving from non-human natural ecosystem processes, while salience of the social system is depicted with increasing dominance to the right. Classifications into cultural services reflect the main interconnections between human and non-human natural ES processes (**Figure 4**).

In terms of number of ES subtypes classifications, fishers and tourism and leisure agents cited a larger array of services (22 and 20 subtypes, respectively), followed by researchers and miners (17 and 15 subtypes). Mariculture and aquatic transport agents displayed a narrower ES subtype classification profile with only nine subtypes.

Fishers were the user group citing more provisioning services of food (subtype 7; n = 58) and genetic resources (subtype 8; n = 55), i.e. they cited many species names for fish, mollusks, and bivalves perceived as benefits from the Babitonga ecosystem. The group of researchers identified services across the range of ES types used in the analysis. Tourism and leisure agents are characterized by a greater reference to ES belonging to cultural subtypes leisure and tourism (subtype 12), legacy and existence (subtype 14), esthetic inspiration and contemplation (subtype 15).

Several ES subtypes are not shared amongst user groups, because they were cited by only a particular user group (**Table 2**). For example, nutrient cycling and climate regulation were cited only by researchers; aquatic transport agents were the only citing a geomorphological resource; miners were the only citing regulation of erosion and hunting; fishers were the only citing spirituality, assistentialism, and funding opportunities and; tourism and leisure agents were the only citing politics as a service obtained from their ecosystem.

On the other hand, our informants perceived several shared services. For instance, food (provisioning), tourism and leisure (cultural), economic viability (e.g. employment, work, and income) and infrastructure/logistics (e.g. transport, vessels, ports, and navigation) (both cultural/people ESs) are shared values by

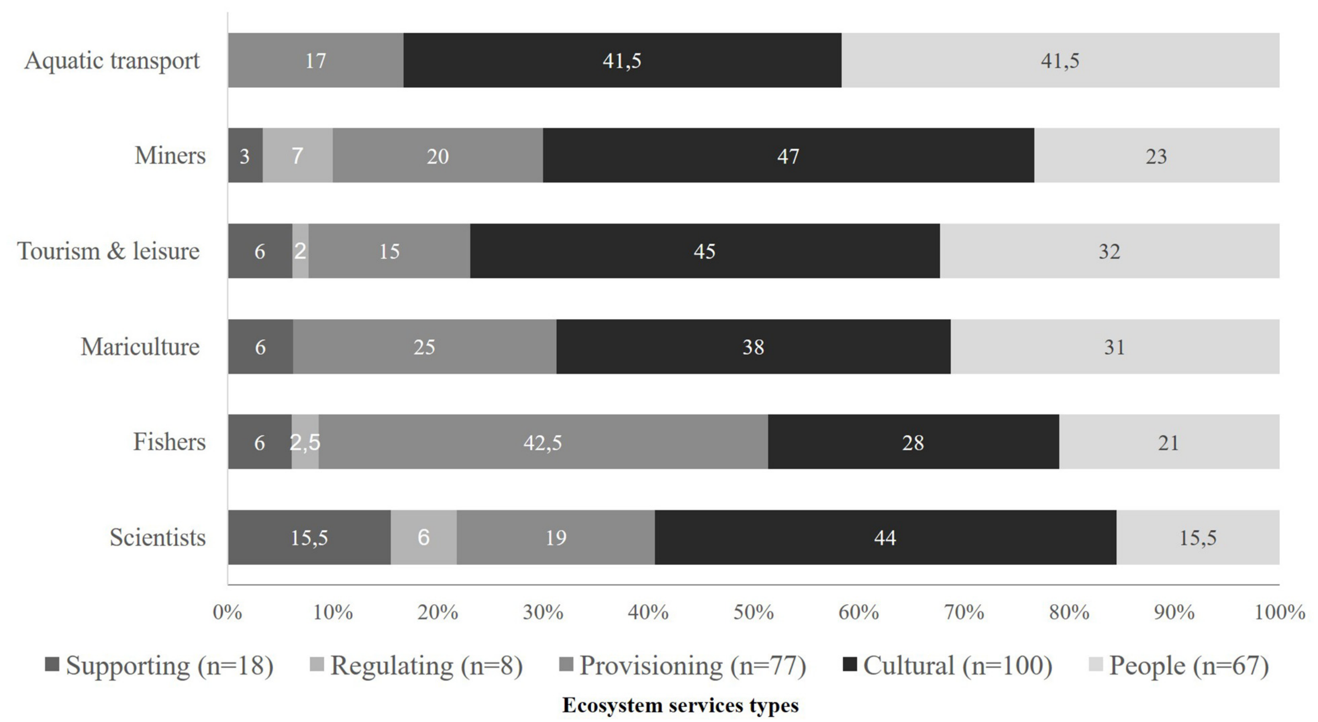

FIGURE 3 | Relative frequency of distribution in classifications of ES types based on the perception of six direct user groups of Babitonga Bay (N = 270 citations), identified in 19 workshops. Numbers given in legends refer to the absolute frequency of classifications per type of ecosystem service.

all user groups. Interestingly, three ES subtypes (maintenance of life cycle; water quality and; cultural and historical patrimony) were mentioned by all user groups, with the exception of aquatic transport agents which were also the only group not citing any supporting nor regulating services.

DISCUSSION

Mapping Patterns in Ecosystem Service Perception Profiles

McNally et al. (2016) observed that different actors tend to assign priorities to ES that are more related to their way of life. Our results outline the structural differences amongst ES profiles perceived by each user group. However, while Hein et al. (2006) hypothesize that local actors would indicate more "provisioning" and "supporting" ES; most of our classifications fell under the categories cultural (62%) and provisioning services (29%).

The ES subtypes we recorded derive from human interactions within the Babitonga Bay environment, where users create and use tools in a cosmological relationship with the natural, non-human components of this ecosystem. Daily cultural practice shapes environmental spaces and are in turn enabled by them generating cultural goods, this whole process enabling cultural ecosystem benefits (Fish et al., 2016). Recent research highlights the importance of cultural services in relation to other ES types (Chan et al., 2012) – since all citizens use and benefit from cultural services, regardless of their economic activity, i.e. leisure, contemplation of the landscape, sense of place, and cultural traditions are largely available to all people, independent of their economic activity.

All ecosystem users in this study valued provisioning services to some extent. But fishers, more than any other group, outstandingly valued this type of ESs through several species of fish mentioned as vivid demonstration of the richness of their local ecological knowledge and ethnotaxonomy of aquatic life. Most provisioning services were either classified as food and/or genetic resources, obtained through commercial or sport fishing activity by most users, and through mariculture activity. Provisioning and cultural ESs are intimately linked, i.e. fishing as a noticeable example has strong bonds with cultural benefits: it can be an economic or recreational activity (Boyd and Banzhaf, 2007); it is a traditional practice enabling a differentiated livelihood; and may be associated with spiritual, therapeutic, feelings of belonging, satisfaction and survival issues. The very existence of provisioning services impels humans to develop cultural structures and practices to extract, plant, and interact with the ecosystem – and when they become scarce we'll see associated changes in cultural practices. In this case, there may be changes in cultural services, and consequently impulse to develop new structures (technologies and constructions) that intensifies or improve the use of provisioning services (cross-ES feedbacks).

Regulating and supporting services were the least mentioned in our study, a pattern also found in other ES perception studies (Raymond et al., 2009; Casado-Arzuaga et al., 2013; McNally et al., 2016). These, ESs were not at all mentioned by

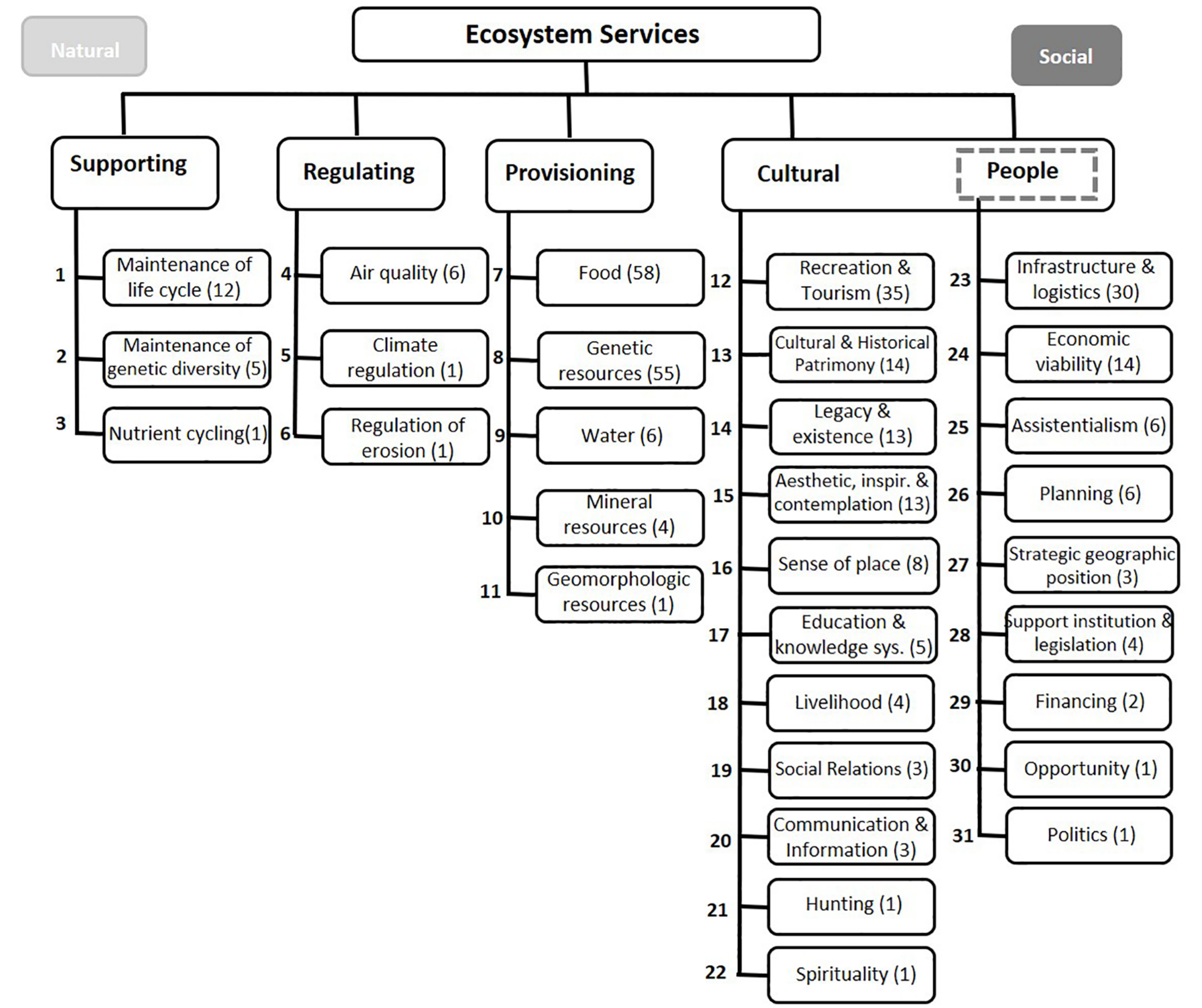

FIGURE 4 | Adaptation of Raymond et al. (2009) ES' *framework* to a social-ecological systems perspective, considering the types of services in a socioecological system gradient, ranging from more natural (supporting, regulating, and provisioning), and social (cultural and people) properties to a main point of connection and interconnection of these characteristics represented by cultural services. We considered 31 subtypes and 317 classifications.

aquatic transport agents – probably because this group work in indoor environments and their economic activity (port and navigation) do not depend directly on the health of the aquatic environment in order to be productive. While this might be a reasonable inference, it does not entirely explain why regulating and supporting ESs were not abundantly cited by other users that have an intimate relationship with the sea such as fishers and mariculture agents. These ES types are often considered indirect benefits (Costanza et al., 1997) and regarded as processes and operating mechanisms of nature; thus not generally noted in perception studies possibly because they are not easily recorded through inductive methods used.

Indeed, Oliveira and Berkes (2014) showed that fishers in Rio de Janeiro do not perceive regulating and supporting services as benefits, but rather as a natural environmental condition. Similarly, it is more evident for people to cite access to clean water as a benefit, than the cleaning process it goes through (Fisher et al., 2009). Therefore, we suggest that such services could be accessed by explicitly probing questions related to specified processes such as climate change (amount of rainfall, drought), water dynamics and flow, role of the mangroves in the ecosystem, and role of different environments in generating life.

Nevertheless, inferences may still be advanced on the variance and similarities amongst ES perception profiles. For instance, we suggest that ESs subtypes cited by only a particular user group, offers an identity marker that differentiate that group and are derived from peculiarities of ES that may define the socioeconomic activity itself. For example, only researchers, who are generally aware of ES and sustainability discussions, referred to nutrient cycling and climate regulation. Similarly, only aquatic transport agents cited the natural depth of channel as ESs because of their dependence on navigation channels to operate large ships. Fishers were the only group concerned with spirituality probably as a reflection of their intimate, direct relationship with the aquatic world.

Our ES perception profiles highlight the benefits that are important for the daily routines and social well-being of all investigated direct ecosystem users and hence to be regarded as shared values. ESs such as provisioning of food by the ecosystem, and cultural benefits such as tourism and leisure, employment, work and income as well as cultural/people's services such as transport, vessels, ports and navigation – should bare special place in the development of sustainability policies. However, our results also show other ESs of critical importance cited by all user groups. The more powerful actors in our study case, the aquatic transport agents, were the only group which did not consider maintenance of life cycle, water quality and cultural and historical patrimony. This may signal lower engagement with issues concerning aquatic ecosystem health.

Implications to Coastal-Marine Ecosystem Service Assessments

Abson et al. (2014) found that the highest percentage of studies in ES were empirical studies of natural science and valuation; and that interdisciplinary studies are still incipient and are mainly related to the dynamics of knowledge systems about services and their political mechanisms. Other studies are overly focusing on monetary values (Richardson et al., 2015), and in many cases, services of extreme importance such as cultural services, are neglected because they are intangible and difficult to assess (Chan et al., 2012). For Jacobs et al. (2016), designing more integrative ES assessment methods has been a pressing but difficult challenge, given usual reliance on varying but hard to conciliate assumptions, axioms and pre-analytical frameworks.

By adopting a deliberative approach using complimentary inductive words (benefits, rights and resources) and accommodating cultural/people's services in our framework, our analysis enabled the integration of informants' own (emic) perspectives of the ecosystem and positioned citizens as both service providers and consumers. ES thus emerged in a real policy-making process as perceptions of complex interactions between the biophysical environment, ecological processes, and human interventions (Mouchet et al., 2014; Bennett et al., 2015).

This study did not adopt the conventional bidirectional model where ecosystem properties or functions and provisioning services are on the supply-side, while sociocultural or social system domain on a demand-side (see Costanza and Folke, 1997; Martín-López et al., 2013; Felipe-Lucia et al., 2015). Our results enacts a conceptual model that regards humans as an integral part of the ecosystem, and not simply an outside force enjoying services produced by nature (**Figure 5**). We thus offer a co-evolutionary gradient from ecosystem processes less-to-more human-agency dominated types of services (following the notion that boundaries between SES are artificial and arbitrary- Berkes and Folke, 1998). We do consider that supporting and regulating services are associated to the biophysical domain, similar to Martín-López et al. (2013), since they exist independently of the human presence in the ecosystem and are basic foundations for the entire natural system. However, our approach differs from the above authors whom placed humans separate to the "ecosystem."

Our model also highlights the existence of feedbacks and trade-offs across the spectrum of ESs rendering further complexity to ES assessments. For instance, the socioeconomic significance of benefits and the meaning people place on the services may have diverse underlying relationships (Oliveira and Berkes, 2014), e.g. they can be classified into multiple types of services as shown in the case of several possible linkages between food provisioning (fish) and diverse possible cultural services immanent in the act of fishing. Human-induced changes in one type or subset of ESs may also trigger cascading effects on the availability of other ESs in the socioecological system gradient (**Figure 5**). For instance, the construction of oyster and mussel aquaculture parks, in a given area, directly engages with environmental features to produce food (provisioning service). While benefits are generated, poor management may cause harmful externalities through pollution by increased organic matter, plastic disposal, and disturbance of traditional navigation pathways. These can in turn affect the capacity of the ecosystem to regulate, support and provide other services, including cultural benefits.

Peterson et al. (2018) have pointed the main advances and shortfalls of the so-called Nature's Contributions to People Framework in relation to conventional ESs approaches (NCP, Díaz et al., 2018). Our ethnoecological lens is highly sensitive to cultural context as a cross-cutting factor shaping human perception of nature and quality of life – which is also a major NCP advancement in the opinion of Peterson et al. (2018). In our opinion, our humans-in SES approach does not emphasize linear or one-directional flows of contributions from nature to people – which is a major shortfall of the NCP according to these authors.

Implications to Coastal-Marine Ecosystem-Based Policymaking

This paper contributes to the "new valuation school" described in Jacobs et al. (2016), by exploring the integration of nature's diverse values in ecosystem-based governance initiatives – when "public goods" (instead of "individualistic preferences") are at stake in coastal-marine policy-building processes. Our research addresses three major features suggested by ES literature for the evolution of integrated valuation (Fischer et al., 2015; Ruckelshaus et al., 2015; Bennett, 2017; Boeraeve et al., 2018; Peterson et al., 2018): (i) inclusive of local/traditional knowledge systems; (ii) based on integrative methods; and (iii) supportive of experimental learning. They particularly concern the inception (early-stage) of ES assessment agendas, i.e. purpose definition and the scoping process (Jacobs et al., 2016). Next, we explore these features on the light of the main science-policy insights gained in the Babitonga study case.

The literature highlights that integrated valuation should (i) use local knowledge systems to enhance research design and improve its societal relevance (inclusionary of hidden values and power asymmetry as part of an iterative science-policy

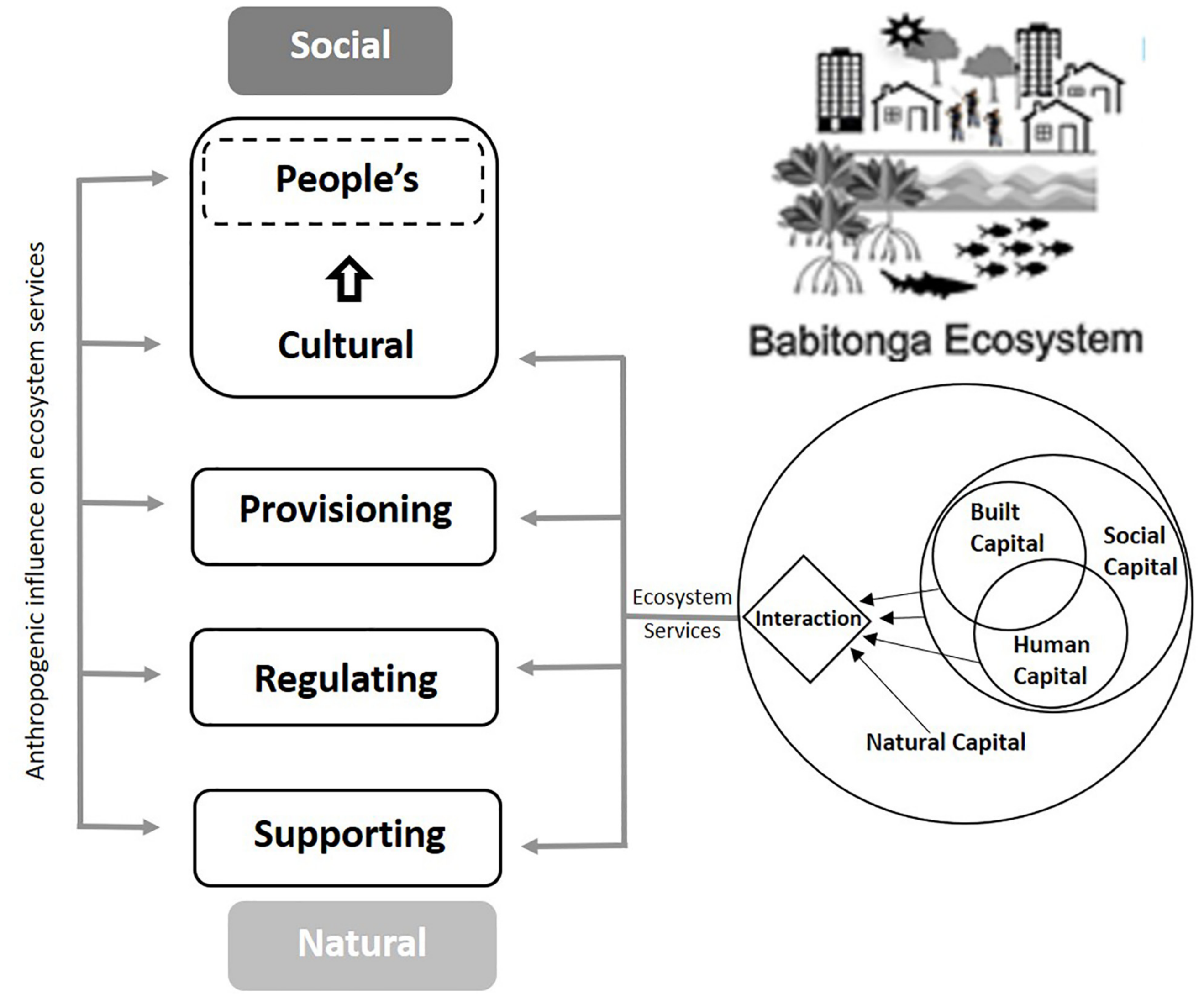

FIGURE 5 | Connections between ES arranged in an interdependent, nested gradient within the focal social-ecological system (e.g. Babitonga Bay ecosystem). We acknowledge that ES as well as complex cascading effects results from the interaction of different types of natural capitals (including non-human derived natural capital, social, human and built capitals; interconnected arrows to the right). Services (etic) or benefits (emic) are perceived by social-ecological system's agents (direct ecosystem users and researchers in the Babitonga Bay ecosystem case study), the structure of which vary from less (Supporting, Regulating, and Provisioning) to more socially dominated (Cultural including People's) types of ES. The interconnected arrows to the left therefore show humans influence on one or more service, not necessarily in one direction, e.g. change in cultural/people's services can influence provisioning and regulating services, and/or all other services in multiple ways) (adapted from Costanza et al., 2014).

process). Our paper describes actors' ES perception diversity, and the implications for developing a territorially bonded "shared values" discourse and practice process. One that is inclusive of ecosystem actors' unique identities and potential contributions, but also embracing a more holistic and inter-dependent view of the ecosystem and its component parts. We noted that perceptions on ES varies according to one's cultural background and, therefore, there is a constant risk of falling into models that privileges the mindsets of those (usually more powerful) humans involved in decision-making. Hence the need to remain watchful and discerning, because power ultimately influences the allocation of and degree to which individuals and groups may be capable of accessing ESs (Felipe-Lucia et al., 2015). Enacting the perceptions of different actors' through deliberative approaches can, therefore, help deepen societal understanding of ecosystem (including cultural) services and steer more equitable management processes (Otero et al., 2013).

Secondly, integrated valuation should (ii) combine methods, disciplines and approaches to enable understanding and thus hopefully increase mutual capacity, ownership, trust, and long-term success. We suggest that the integrative nature of ES assessments approaches calls deliberative methods, because integration will most effectively emerge naturally through the realization of the place and role of each other actor group in the future making of the SES. Our ES perception profiles may become a valuable social learning tool because they help contextualize the interplay between ecological knowledge and power in policy making turning the realization of these relationships more explicit in deliberative processes. For instance, some patterns across the spectrum of ES perception profiles, when brought to the table and discussed by resource users, will be seen as proxies of potential conflicts or divergence of expectations in terms of future visions for the SES.

Our results therefore set higher standards for upcoming blue economy debates in Babitonga Bay and across Brazil. They will thus hopefully challenge neoclassical monetary valuations, individualistic non-monetary approaches, thus helping to avoid development of non-monetary/socio-cultural valuation as a separate research domain (Kenter, 2016). Conventional economic thinking narrows its very definition of value to elements people perceive as direct benefit and are willing to pay for (Costanza et al., 2017). These are predominant approaches in ES studies, which can result in several key ES ignored and/or undervalued, incentivizing policies to maximize a select few services ("cherry-picking") based on data availability and ease of quantification (McDonough et al., 2017) – with consequent socially and ecologically undesired effects (Kull et al., 2015).

Finally, integrated valuation should also (iii) enable reflexivity and experimentation through sets of new scientific parameters for future policy evaluation. Our research is embedded in a "transformations in the making" SES opportunity context at the Babitonga Bay ecosystem level (Gerhardinger et al., 2018b). While our workshop participants are slowly becoming aware and engaged in the reflection about and uptake of the data generated by each cycle of participatory planning series, the results presented in this paper already places us (researchers) in a much better position to represent their values, worldviews and expectations in transformative policy making codesign. In this regard, Gerhardinger et al. (2018b) application of Westley's et al. (2013) TAT provides us specific-phase recommendations of institutional entrepreneurship strategies, skills, actions and types of agency required for fulfilling the vision of and navigating toward an ecosystem-based governance regime at Babitonga Bay ecosystem. TAT tells us it is critical to encourage the proliferation of ideas and the recombination of resources in new forms (e.g. building networks, making room for desirable emergent self-organization); that we should help a new dominant design to emerge by encouraging the dropping off of some ideas and linking those that are agreed offer a viable alternative platform and; that we should enable resource mobilization through leveraging and brokering (e.g. identifying opportunities, engaging the emerging energy of the system, working through networks and partnerships, connecting ideas and resources). What these prescriptions means in practice?

Paramount to our on-going transformation is for research-action projects to continue creating room for a more diverse ES perception base to confront current dominant views of Babitonga's vocation for ports. Envisioning a more diverse identity for this SES where all ecosystem actors can prosper is perhaps the key desirable idea to inspire future social learning. For instance, empowering less powerful and hence represented groups in territorial development policies, such as fishers, mariculture, tourism and leisure agents, should be regarded as priority targets by external agents willing to support their collective action and political organization. Given the lack of socio-political organization these groups are known for locally, strategies such as citizen-science and self-monitoring the health and productivity of the aquatic environment seems to be good starting points – to connect their experiential knowledge of the aquatic ecosystem through evidence-based agendas will enact their authority in the operations of new knowledge-building, problem-solving and decision-making stances (such as the emerging PBG multi-stakeholder platform). This is where an important aggregate of shared values discourse made explicit through our results meets practice, with the potential to frame the terms for future ecologic-economic zoning discussions in Babitonga Bay.

Timing is critical here because in the upcoming years, the collective action energy of less influential actors could be fully drawn to a reactive agenda, i.e. if massive dredging operations are authorized by the triggering of the installation phase of new ports and a shipyard, the quality of the water may immediately drop and severely affect fishing and aquaculture operations (Gerhardinger et al., 2018a). For instance, fishers are facing the risk of not being able to maintain the very own existence of artisanal fisheries as a viable activity. Unfortunately, this is not an isolated circumstance, but a widespread example of the unfair trade-offs effects generated by fragmented licensing of coastal infrastructure (e.g. new ports), exacerbated by the greater social and political vulnerability and marginalization of small-scale fisheries in Brazilian developmental policies (International Collective in Support of Fish Workers [ICSF], 2016).

CONCLUSION

Our analysis demonstrates that even before the criticisms on the use of the word "benefit" in the definition of ESs (a synonym of ES to some), it was capable of eliciting the essence of ES from different direct ecosystem actors' perspective. Our integrative and deliberative approach encompassed, in addition, the words "rights" and "resources" thus broadening the diversity of typologies assessed and required consideration by the political system in governance and territorial development initiatives. Since ES is an academic-scientific definition to be used in management processes and public policies, researchers need to be aware of its limitations when conducting research involving different social actors. Thus, we argue that the formal definition of ES should be broadened to consider a wider range of services than what is currently contemplated in conventional ES studies, such as "benefits produced and obtained within the socioecological system." This is a fundamental notion since humans can both use and produce ESs, as well as positively and negatively influence its availability and quality.

Our paper also reinforces the importance of cultural services, because regardless of the economic activity performed, every citizen benefit from them even though they are rarely properly valued and considered in management and development.

The overvaluing of a specific subset of ES, usually associated with the interests of a smaller and more empowered social group, is among the main causes of civilizational crises. ES studies thus have the noble and challenging role of imbuing collaborative and integrated strategies of territorial planning with greater distributional justice. This could be achieved through valuation strategies capable of building alternative visions for sustainability that are based on values that are shared amongst actors, but also sensitive to the identities of more vulnerable stakeholders.

Our results therefore seriously challenge dominant patterns of neoliberal styles of planning by exploring a scalable and replicable approach to symmetrically contextualize in marine policy, the structure of perceived services by a wide range of economic agents – from more powerful (mining and transport agents) to less influential (small-scale fisheries and mariculture). We set new terms for strategic, hopefully transformative, social learning to take place; by translating the diversity of direct ecosystem users' perceptions into a more coherent and integrated approach to ES that may hopefully lead toward more inclusive, equitable and ecocentric policymaking of disputed seascapes.

ETHICS STATEMENT

This research was approved by the Ethics Committee of Federal University of Santa Catarina (CAAE 42938115.1.0000.0118).

AUTHOR CONTRIBUTIONS

DH, LG, and NH designed workshop methodology and wrote the manuscript. DH and LG performed the workshops. DH analyzed the datas.

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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.2020.00083/full#supplementary-material

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