Guiding Insurance Instruments to Leverage Natural Infrastructure for Climate Change Resilience

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March 2019

A Group Project report submitted in partial fulfillment of the requirements of the degree of Master of Environmental Science and Management for the Bren School of Environmental Science & Management, University of California, Santa Barbara.

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As authors of this Group Project report, we archive this report on the Bren School's website such that the results of our research are available for all to read. Our signatures on the document signify our joint responsibility to fulfill the archiving standards set by the Bren School of Environmental Science & Management.

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3/22/2019

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Acknowledgements

We would like to extend our gratitude to all who contributed to our project and provided their support.

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Dr. Allison Horst, UCSB Bren School
Dr. Siddharth Narayan, UC Santa Cruz
Mark Way, The Nature Conservancy

This research is funded [in part] by Chubb.

Abstract

Climate change poses a substantial threat to coastal cities through sea level rise and storm surge. Natural methods of coastal protection, such as coral reef restoration, are increasingly implemented to mitigate storm damage. Coral reefs attenuate wave energy by up to 97 percent and provide additional benefits, or co-benefits, such as ecotourism and erosion control. The Nature Conservancy has explored the risk reduction benefits provided by coral reefs; however, governments and environmental managers do not currently have a framework for quantifying the co-benefits of coral reefs. In order to encourage investment in coral reef restoration and to inform future restoration plans, we i) developed a universal approach to value coral reef co-benefits, ii) built a model to more effectively gather data on the cost of coral restoration projects, and iii) determined broad feasibility of Caribbean coral restoration in the face of climate change given the current location of coral suitable habitat. The results of our three objectives may be used to incentivize coastal property owners to invest in coral reef restoration not only for storm protection, but also for the ecotourism and beach preservation benefits coral reefs provide.

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Executive Summary

Introduction

Current approximations determine that 44% of the world's population lives near coastal areas that are at high risk from storm and sea impacts (Rahmstorfa 2017). Mitigation of impacts for both coastal populations and the properties on these coasts has been a growing field of interest for international researchers, non-governmental organizations, and governing bodies. A wide variety of approaches to manage these impacts have been explored, such as the construction of manmade "grey" infrastructure (seawalls, breakwaters, artificial reefs, etc.) or sand replenishment. Researchers have also explored the potential that "green" or natural infrastructure (coral reefs, mangroves, etc.) has for wave attenuation and storm mitigation in addition to or in lieu of grey infrastructure. Apart from these coastal protection benefits, research exploring associated benefits, or co-benefits, of natural infrastructure (e.g., ecotourism and aesthetic value) has grown over the past few decades. However, many coastal habitats have either deteriorated or been completely lost due to human development and climate change. As a result, multiple governing bodies and organizations have invested substantial funding and resources to restore coastal habitats.

Motivated by the potential direct and indirect benefits of natural infrastructure and considering restoration costs, The Nature Conservancy (TNC) and other research entities such as the University of California, Santa Cruz (UCSC) and the University of Cantabria (UC) have spent the past decade quantifying the wave and wind reduction benefits natural ecosystems provide for coasts and have begun to explore creative funding mechanisms for restoring these ecosystems to promote coastal resilience. One approach considers potential insurance plans for coastal properties (hotels, condos, etc.) in high tourist areas that leverage incentives to mitigate storm risks primarily through the valued direct and indirect co-benefits coral reefs provide.

To help incentivize hotels and insurance companies to invest in reef restoration, our team established three main objectives. These objectives focus on i) developing a universal approach to value coral reef co-benefits to hotels, ii) building a model to gather more complete data on the cost of coral restoration projects, and iii) determining feasibility of Caribbean coral restoration in the face of climate change given the current location of coral suitable habitat.

Methodology

Objective 1: Develop Replicable Approach for Valuing Coral Reef Co-Benefits

As ecotourism and beach erosion control are important drivers of hotel revenue in the Caribbean, we decided to quantify these two co-benefits of coral reefs. We quantified the value of ecotourism, measured by coral cover and distance to the reef, and erosion control, measured by beach width, to hotels using the hedonic pricing technique. This method uses regression analysis to isolate the contribution of specific variables to the composite price. Using the data for 372 hotels in 5 Southeast Florida cities collected via TripAdvisor and ArcGIS, we performed a regression analysis to determine the influence of beach width, coral cover, and distance to the reef on the nightly price of a standard two-person hotel room. This method can be applied to any region as long as there is sufficient data on coral cover and hotel amenities.

Objective 2: Determine Grey and Natural Infrastructure Cost

After an extensive review of the literature revealed a large gap in data around restoration cost, we conducted phone interviews with experts in the field to gather qualitative data on why restoration cost is so variable and difficult to aggregate and quantitative data on restoration cost separated by the various project components and stages.

Objective 3: Identify Future Suitable Habitat for Coral Restoration

In order to determine the feasibility of Caribbean coral restoration in the face of climate change, our team conducted a coarse species distribution analysis using Maxent. We began with a literature search to identify the main environmental features important for coral survival in the Caribbean, which include sea surface temperature, pH, salinity, photosynthetically active radiation, and ocean current speed. We then identified the spatial distributions for four families of reef-building corals (*Acroporidae, Montastraeidae, Merulinidae,* and *Poritidae*). Next, we used the Community Earth System Model (CESM), a global climate model that provides computer simulations of the Earth's past, present, and future climate, to subset three time periods with varying biogeographic conditions of the important environmental variables; specifically current (2008 - 2018), near-future (2040 - 2050), and future (2070 - 2090). These three time periods were taken from two CESM climate scenarios: RCP8.5 (business as usual) and RCP4.5 (curbed greenhouse gas emissions).

We combined the environmental layers and species presence points in Maxent, a program that merges coral presence points with varying environmental layers to determine environmental niches and model current and future species habitat suitability. The resulting gains and losses in coral suitability across the two climate scenarios and three time steps were mapped in ArcGIS.

Results

Objective 1: Develop Replicable Approach for Valuing Coral Reef Co-Benefits

Our regression analysis revealed that both coral cover and beach width are significantly positively correlated with hotel price and that this relationship depends on the region. Using the coefficients on the variables of coral cover and beach width as the willingness to pay for an additional unit of coral cover or beach width, we were able to extrapolate to the total annual willingness to pay for these variables. For example, in Fort Lauderdale, our research indicates an additional 1 km^2 of coral reef is worth \$35.14 million and an additional 1 m of beach is worth

$1.28 million to hotels. Our research also suggests this method is most appropriate to apply in regions that are more dependent on reef tourism. These economic values can be added to the protection value of reefs found by TNC to provide potential investors with more accurate valuations of the total benefits provided by coral reefs.

Objective 2: Determine Grey and Natural Infrastructure Cost

Interviews with experts in the field of reef restoration revealed the many components and stages that are involved in reef restoration and the factors that influence the variability of cost by location, organization, and growing technique. We found a range of estimates of $60-90 per m^2 for coral restoration cost in Miami, Florida, where the majority of our interviewees work. Additionally, we contacted the Army Corps of Engineers (Corps), who are responsible for funding and constructing some of the larger grey infrastructure projects along the US coastline. The Corps out of Jacksonville, Florida reported an estimate of $6,234 per linear meter of vinyl sheet pile and $7,874 per linear meter of steel sheet pile. Both our literature review and expert interviews revealed that the cost of restoration is substantially lower than the cost of artificial infrastructure, such as seawalls and submerged breakwaters. To facilitate the calculation of a project's total cost, we developed an Excel model that restoration groups can use to input and aggregate their individual costs to construct a total budget for a project. This enables restoration groups to more readily determine and report the cost of specific projects based on location and technique.

Objective 3: Identify Future Suitable Habitat for Coral Restoration

We found that future coral reef habitat does not experience an absolute loss, rather an overall shift in both climate scenarios and three time periods. Modeled gains in suitability were limited to the Florida Reef Tract and the Mesoamerican Barrier Reef System. The suitability gains are restricted to the southern end of the Florida Reef Tract and Mexico's Yucatan Peninsula, Belize, and Guatemala. We found that sea surface temperature, salinity, and ocean current speeds collectively contribute the most to coral suitability, with photosynthetically active radiation (PAR) and pH contributing the least. This analysis can be used to prioritize restoration efforts or further assess coral health and threats in Caribbean regions with predicted high suitability.

Conclusions

Coral reefs are much more than an important natural defense against storms; they also provide valuable co-benefits, a portion of which is directly captured by coastal hotels in the form of increased revenue. Our valuation of co-benefits can be used together with risk reduction data from TNC to enable hotels to make informed investment choices in preparation for a changing climate and more frequent and severe storms. The cost model we developed will help hotels project financial return on investments in coral reefs, which is a vital step in incentivizing restoration projects. Finally, our suitability maps will allow TNC to make strategic decisions to ensure restoration projects implemented today will be successful in the future. Together, our co-benefit valuation approach, cost model, and suitability analysis will allow TNC to identify areas with promising investment opportunities for hotels that may result in coral reef restoration via an insurance mechanism.

Project Significance

Previous studies have shown a connection between coral reefs and coastal resilience, and The Nature Conservancy is using this research to motivate new and more substantial sources of funding for reef restoration projects. As coastal storms and flooding become more frequent, properties along the coast face an increased risk of damage. Since reefs are now considered a protective barrier along the coast, TNC is leveraging restoration as a method to better protect coastal properties. As coral reefs face many threats with the changing climate, continuous funding is required to rehabilitate and maintain these important and threatened ecosystems along the coasts. Fortunately, the field of restoration is growing as governments recognize that many different sectors would benefit from nature-based infrastructure such as coral reefs.

To help incentivize both hotels and insurance companies to contribute to reef restoration, we first determined and valued the additional benefits of coral reefs that can be added to the direct flood reduction benefits that reefs provide. We gathered an estimate of costs associated with restoration and grey infrastructure, and learned which key factors make up the cost of a restoration project. We analyzed the costs and co-benefits that hoteliers could expect from investing in coral reefs, and provided a comparison to the costs of equivalent grey infrastructure, such as submerged breakwaters and levees. Additionally, we conducted a feasibility analysis to determine where suitable habitat may exist for coral in future years, which could provide The Nature Conservancy with recommendations on where to choose case study sites to apply our cost and benefit methodologies.

We focused our project on the third largest barrier reef ecosystem in the world, the Florida Reef Tract. This reef ranges from the Dry Tortugas to Biscayne Bay and provides important wave attenuation and ecosystem services to the Florida population and economy. The criteria for choosing southeast Florida as our case study site are (1) available data on co-benefits, (2) natural and grey infrastructure present, (3) tourism appeal, (4) coral restoration prevalence, and (5) storm and flooding threats. However, climate change and associated increasing storm events threaten the regions, and efforts to restore natural infrastructure are increasing. An insurance policy for natural infrastructure in these regions, proposed by The Nature Conservancy, will help fund these ongoing restoration projects and benefit Florida’s economy. Our research assesses the feasibility of using hotel insurance premium discounts to fund the maintenance and restoration of local natural infrastructure.

Background

Introduction

Climate change poses a major threat to coastal cities through sea level rise and storm surge. Sea level rise is attributed to two primary drivers: (1) thermal expansion of seawater due to increasing ocean temperatures and (2) water input from the melting of land ice (Nicholls and Cazenave 2010). When only considering sea level rise, this phenomenon is expected to have adverse effects on coastal zones. Short-term effects include coastal flooding and saltwater intrusion of surface waters, while long-term effects include increased erosion rates and saltwater intrusion of groundwater (Nicholls and Cazenave 2010). The combination of sea level rise and climate change are expected to further influence extreme events such as storm and flood events. Warming sea surface temperatures are a critical driver of altering wave climates by influencing wind and storm patterns (Reguero et al. 2019). Sea level rise synergistically interacts with wave energy through changing wave dynamics and increasing wave propagation which can intensify effects such as beach erosion (Albert et al. 2016). These processes are predicted to impact coastal populations globally.

It is estimated that 10% of the world's population lives on the coast; however, a large proportion of coastal communities are poorly constructed on improper foundations or insufficient land planning, making these towns particularly vulnerable to the effects of climate change (Spalding et al. 2014; McGranahan et al. 2007). Furthermore, up to 1.2 billion km^2 of land, 310 million inhabitants, and US$11 trillion dollars of built capital are located within the threshold of a 100-year flooding event (Reguero et al. 2015; Hinkel et al. 2013; Neumann et al 2015). A 100-year flood or 100-year storm event refers to a large flood or storm with a 1% statistical probability of occurring (Perlman 2017). As storms increase in intensity and frequency, damages to coastal towns will rise. Increasing wave energy and storm prevalence results in higher rates of erosion and damage to coastal properties, requiring coastal cities to invest in structural barriers such as breakwaters and seawalls (Spalding et al. 2014). These actions are often an emergency response to critical erosion rates or protective efforts for upcoming storm seasons. Governments have spent billions of dollars on reactive measures such as implementing grey infrastructure (i.e. submerged breakwaters, seawalls, levees) against climate change. As a result, there is currently a strong effort to identify cost-effective and proactive management strategies for climate change mitigation, leading researchers to consider nature-based solutions such as coral reefs (Emillson and Sang 2017).

Coral reefs are extremely important marine ecosystems that provide myriad ecosystem services to the global economy through direct use, indirect use, and non-use services (Table 1). They are found in coastal tropical environments from 25°S to 25°N and exist in the top 100m of the water column (Hoegh-Guldberg 1999). Apart from the coastal protection provided by coral reefs, coastal communities also benefit from corals through ecotourism, recreation, fishing, sand stabilization, research, and education (Deloitte Access Economics 2017). People visit tropical locations annually to participate in reef activities such as snorkeling, scuba diving, glass-bottom boats, and fishing trips, among other recreational activities. In recent years, there has been a strong emphasis on researching the benefits coral reefs provide to the global economy.

Table 1. Co-benefits of coral reefs (Deloitte Access Economics 2017; Burke et al. 2008; Moberg & Folke 1999). This list comprises of benefits to communities as a whole.

Ecosystem Service Classification	**Coral Reef Services**
Direct Use (Provisioning)	• Commercial & subsistence fishing • Eco-tourism & educational value • Recreation • Genetic resources • Pharmaceuticals • Ornamental resources • Building materials (lime, mortar, and cement)
Indirect Use (Regulation & Maintenance)	• Erosion control • Sand formation • Storm protection • Nutrient cycling
Non-Use (Cultural)	• Spiritual and religious values • Cultural significance • Intrinsic value

Coastal Protection by Coral Reefs

Scientists at The Nature Conservancy (TNC) and elsewhere have heavily researched the coastal protection services that natural features such as coral reefs, wetlands and oyster reefs provide. While they vary in biological and geographic characteristics, they all act as a defensive barrier from the ocean either by reducing wave energy through bottom and drag friction, as well as depth change (Narayan et al. 2016). Ferrario et al. (2014) determined coral reefs can reduce wave energy by up to 97%. They act as “natural, low-crested, submerged breakwaters” providing storm surge and flood reduction benefits to adjacent shorelines (Beck et al. 2018). Reef characteristics such as roughness, water depth, and coral cover determine the magnitude of wave attenuation by transferring wave energy from the sea surface to the reef itself (Spalding et al. 2014; van Zanten et al. 2014). However, while these protection services are significant, they are often not realized by the general public.

In order to translate the effectiveness of nature-based solutions to the non-scientific community, TNC has also conducted extensive research on monetizing the direct benefits coral reefs provide to our economy. To value the global flood protection savings provided by corals, Beck et al.

(2018) evaluated avoided flood damages to built coastal capital using historical and modelled datasets describing hydrodynamic conditions. They estimated that coral reefs protect approximately 258,000 people, US$6 billion dollars of built capital, and 470 km^2 of land from flood damages (Beck et al. 2018).

TNC also developed an approach to value coral reef ecotourism and recreation. Spalding et al. (2017) designated "on-reef" (e.g. diving and snorkeling) and "reef-adjacent" (e.g. reef views and fresh seafood) activities to capture the value of recreational and touristic benefits that corals provide. Using data from the UN World Travel Organisation (UNTWO) and Global Accommodation Reference Database (GARD), Flickr images, and diveboard.com, coral reefs were estimated to contribute US$35.8 billion dollars to the global economy annually through reef tourism (Spalding et al. 2017). As part of this study, TNC partnered with Microsoft to use social media and artificial intelligence (AI) to detect travel specifically for coral reef tourism. Given the large benefits coral reefs provide to our economy, coral reef cover and health have recently become a priority for tropical countries. However, this approach depends on travel to and social media use at the majority of countries with coral reefs. Many countries may not have infrastructure that supports frequent social media use; therefore, this approach may not entirely capture the value of recreational and tourism benefits.

Coral Restoration in a Changing Climate

Coral reef restoration is an increasingly popular industry for maintaining or recovering coral populations. Hundreds of non-governmental organizations are involved in coral restoration projects using varying restoration techniques. Effective reef restoration includes transplantation of coral polyps from coral nurseries or donor colonies to coastal substrates (Bayraktarov et al. 2016). However, many of these restoration projects have typically been conducted with the purpose of conservation. In an effort to construct reef structures that maximize coastal protection and economic benefits, TNC has placed an emphasis on hybrid restoration techniques containing both green (e.g. coral) and grey (e.g. concrete) infrastructure. These techniques consist of transplanting coral polyps to submerged breakwaters, providing both protection and ecological benefits of a natural reef (Reguero et al. 2018b). Hybrid reefs also ensure protection against strong storms in the event of direct damage to the coral. All coral restoration techniques require a significant amount of time to provide protection and ecological benefits, as corals grow between 2-10 cm per year (Barnes 1987). Hybrid restoration ensures immediate protection benefits via grey infrastructure and gradual ecological benefits, compared to coral transplantation to coastal hard-bottom which will provide both benefits gradually. Therefore, restoring reefs with these two benefits in mind is crucial for planning for climate change mitigation.

However, climate change also threatens the distribution of coral reefs globally. Changes in climatic conditions are leading to shifts in species ranges and increasing extinction rates

(Freeman et al. 2013). Coral reef degradation is driven by both local and global factors (Bruno and Valdivia 2016); these threats include overfishing, eutrophication, coastal pollution, sedimentation, anthropogenic impacts, elevated sea temperature, UV radiation, and ocean acidification. Coral bleaching is a temporary stress-response to extended warm periods in which the corals expel their algal-symbiont, zooxanthellae, which provides them with oxygen, nitrogen, and food. The frequency and intensity of bleaching events are expected to increase as sea surface temperatures continue to rise (Ainsworth et al. 2016). Corals are particularly vulnerable to climate change due to the tendency of reef-building corals, such as *Acropora* species, to settle in their upper thermal limits (Winkler et al. 2015). With the current warming trajectory, referred to as "RCP 8.5", coral thermal limits will be surpassed in the next 100 years in tropical regions without acclimation or adaptation to reduce the severity of coral bleaching and ameliorate mortality (Selig et al. 2012; Ainsworth et al. 2016). Given this trajectory, important aspects of reef restoration include identifying resilient coral genotypes and considering climate projections to guide restoration projects in future viable habitats.

Project Scope

As climate change impacts progress, the size and magnitude of restoration projects will need to increase in order to maintain the significant ecological benefits and coastal protection services coral reefs provide. This upscaling of restoration will require additional sources of funding. Companies are beginning to engage in conservation investing, a growing field aiming to achieve environmental protection objectives while increasing revenue (Reed 2013). Stakeholders such as governments, insurance companies, and public and private companies are affected by climate change and can be involved in conservation investing. In this analysis, we focus on hotels as our primary stakeholder. Hotels are a large private industry which dominates the coastlines of tropical regions and receives the most benefit from a public feature: coral reefs. In other words, hotels with nearby coral reefs are particularly good candidates for engaging in conservation investing due to the storm protection, ecotourism, and erosion control benefits coral reefs provide.

TNC is currently partnering with the state government, hotel owners, and local community in Mexico's Quintana Roo region to develop the "Coastal Zone Management Trust" (The Nature Conservancy, Winters and Festa 2018). This trust receives taxes paid by the tourism industry and local governments to fund coral reef restoration, acting as an insurance policy on the adjacent coral reefs. Triggered by strong wind speeds, the insurance policy pays out to support restoration if there is a certain amount of damage to the reef (The Nature Conservancy). This type of insurance is a reactive measure to direct reef damage involving fixed costs from the tourism industry.

Additionally, TNC is developing a coral resilience insurance policy which proactively restores corals to protect the coast. This policy will insure hotel buildings against flood damage through hybrid reef restoration. In comparison to the "Coastal Zone Management Trust," hybrid green-gray reefs will be established pre-storm season instead of restoring coral reefs post-storm. As the reef grows to provide substantial coastal protection, hotel insurance premiums will be lowered. Quantifying the additional benefits provided by coral reefs is needed to design insurance policies and improve valuation of hotel insurance premiums.

Funding nature-based solutions through insurance policies has the potential to provide a more cost-effective means to protect coastlines as opposed to grey infrastructure. Previous studies determined that other natural defenses such as saltmarsh restoration were three times cheaper than implementing a breakwater (Narayan et al. 2016). However, there are gaps in knowledge on the added benefits, henceforth "co-benefits," that coral reefs provide to hotels, in addition to the costs of restoration projections. Specifically, (1) what co-benefits corals provide to hotels, (2) how to quantify these co-benefits, and (3) average costs of coral restoration projects. Knowledge of these aspects will help stakeholders assess the value of investing in coral reefs. TNC has asked NaturallyInsured to identify and value the co-benefits that corals provide to hotels beyond risk reduction in order to evaluate the cost-effectiveness of coral restoration compared to grey infrastructure.

In Chapter 1, we identified the co-benefits corals provide to hotels and developed an approach to value them, while taking into account data limitations on coral characteristics and restoration costs. The goal of developing this approach was for TNC to easily apply our method to any region of interest given data availability. Through an extensive literature search, we specified erosion control and ecotourism as the most significant and quantifiable co-benefits that coral reefs provide to hotels. We quantified each co-benefit using the hedonic pricing technique, by estimating the relationship between hotel room price and independent variables such as beach width, coral cover, and distance to the reef (Wielgus et al. 2010). In this chapter, we further discuss our reasoning for determining the important co-benefits, variables of interest, and benefit valuation technique used. We then applied this approach to a case-study in Florida.

Next, we discuss a cost model we developed to calculate the total cost of a restoration project based on various inputs in Chapter 2. The goal of this cost model is to streamline all of the different aspects of coral restoration projects to increase future reporting on individual and total costs. This model will help TNC build on their current research to assess the most cost-effective options for coastal protection.

Finally, in Chapter 3, we assess the feasibility of restoration projects while taking climate change into consideration. Through literature review, we determined that there is no one approach for coral restoration due to the variability in biogeographic and economic factors in regions with corals. Because hybrid green-grey restoration techniques require at least 10 years for the coral to grow and provide additional protection benefits, identifying regions conducive to coral growth is important for implementing successful restoration projects and minimizing spending. In this chapter, priority restoration sites were determined via predicting suitable habitat for coral reefs in the future under various climate scenarios.

Project Objectives

Naturally Insured's main objectives are as follows:

1. Create an approach that can be applied to any region to value the beach erosion control and ecotourism benefits of coral reefs to hotels.

The first objective of this project aims to develop an approach for measuring the benefit of erosion control through beach width and the benefit of ecotourism through distance to reef and coral cover. Our method allows TNC to apply this approach to virtually any location, assuming there is sufficient data on coral reef distribution and hotel amenities.

2. Identify the cost of coral reef restoration.

Our second objective aims to more accurately determine the cost of coral restoration and document the factors that account for the high variability in cost. Hotels cannot plan or project financial return on investments without having more complete information on restoration cost. Our model will provide this cost information to hotels to encourage investments in coral reefs.

3. Model coral suitable habitat in the Caribbean and Florida under two climate scenarios (RCP 8.5 and RCP 4.5) using species distribution modeling to determine how corals will shift with climate change.

With climate change, coral reef distributions are expected to shift to more conducive habitats. The final objective of this project aims to determine where coral habitat will be located in 2050 and 2080, which will inform the most viable locations for future restoration plans.

Objective 1. Develop Approach for Valuing Coral Reef Co-Benefits

Introduction

Our team's first objective was to create a method for valuing the coral reef co-benefits of ecotourism and beach erosion control to hotels. This method is designed to be applied in any region as long as there is available data on coral cover and hotel amenities.

Identifying Coral Reef Co-Benefits

Corals and other organisms that inhabit coral reefs supply sand to adjacent beaches. Further, corals control rates of beach erosion by reducing the energy of incoming waves (Guannel et al. 2016; Wielgus et al. 2010; Wells and Ravilious 2006). The team built upon a study by Wielgus et al. (2010), which valued erosion control benefits to hotels in the Dominican Republic. Wielgus et al. (2010) estimated that resorts will lose $52-100 million in revenue over the next 10 years due to beach erosion attributed to coral reef degradation. Wielgus et al. (2010) investigated the capacity for dead, disintegrating coral reefs to protect coastlines; they found that beach erosion rates in the Dominican Republic could increase by more than 80% if the remaining live coral is lost. According to a survey conducted at airports in the Dominican Republic, 25% of visitors noted "beach quality" as the main reason for visiting the country, second only to "climate" (Wielgus et al. 2010). Tourists visit the Caribbean not only to enjoy wide beaches, but also to recreate at nearby coral reefs.

A previous study conducted by TNC valued ecotourism by measuring the monetary value of "on-reef" and "reef-adjacent" activities through social media (Spalding et al. 2017). Spalding et al. (2017) considered activities indirectly linked to reefs, such as enjoying clear, calm waters and outstanding views. Including these "reef-adjacent" activities in our study would require determining the importance of these factors to hotel guests, which we did not have the time or resources to complete (Table 2). The approach outlined in Spalding et al. (2017) is limited because it does not directly measure coral characteristics, such as area and health. In addition, it does not capture the total percentage of tourists due to its requirement of social media use and cell service at these destinations. Therefore, to value ecotourism, we measured the variables of coral cover and distance to the reef.

We chose these three variables as a metric for valuing ecotourism under the assumptions that: 1) guests are more likely to want to dive in reefs with high coral cover, 2) guests care about the proximity of the hotel to the coral reef, and 3) guests enjoy diving, snorkeling, fishing, or boating on the reef.

Identifying Economic Valuation Method

We conducted a literature review of economic valuation methods currently being used to value co-benefits. Some of these methods used in past studies include 1) travel cost, 2) contingent valuation surveys, 3) benefits transfer, and 4) hedonic pricing. The first method uses the cost of travel to and from the ecosystem to assess recreational value. The second values ecosystem goods and services based on stated willingness to pay, and the third transfers valuation data from studies already conducted in other sites or contexts. A substantial amount of time and resources are needed to apply the first two methods, so we did not consider travel cost or contingent valuation feasible for this study. As the third method requires close correspondence between study site(s) and does not apply a generalizable approach that can be used at all sites, we decided benefits transfer was not appropriate for our objective. Thus, we chose to apply the hedonic pricing method.

Table 2. Summary of the four economic valuation methods we considered to quantify coral reef co-benefits.

Methodology	**Description**	**Pros**	**Cons**
Travel Cost	Uses travel cost to and from the ecosystem to assess recreational value	Most commonly applied revealed preference, non-market valuation technique	Assumes all travel costs are incurred to access one site; may only be appropriate for domestic tourism; may not take into account multi-purpose trips
Contingent Valuation Surveys	Values ecosystem goods/services based on stated willingness to pay	Most robust method to value domestic non-use values	Vulnerable to bias and requires careful survey design; expensive and difficult to replicate
Benefits transfer	Transfers valuation data from studies already conducted in other sites or contexts	Does not require time consuming valuation study. Some data is easy to transfer.	Requires close correspondence between study site(s) and policy site(s). Only as accurate as the original study.
Hedonic Pricing	Regression analysis to evaluate how ecosystems influence market prices	Once data is acquired, quick evaluation	Need a lot of data to ensure capturing all factors and no biases

Hedonic pricing is a revealed preference method that studies how different characteristics of a good, both intrinsic qualities and external factors, influence the market price (Weber 2014). In environmental economic valuation, the hedonic method is used to study how varying levels of an environmental good affect the price (Weber 2014). Hedonic pricing uses regression analysis to value the environmental attribute while controlling for other factors (Weber 2014). This method is typically used in real-estate or wage models, but has also been applied to hotel prices (Rigall-I-Torrent et al. 2011, Latinopoulos 2018, Soler et al. 2019).

Advantages of the hedonic pricing method include its versatility to model different relationships and use of existing market data. Another strength of this method is that value is estimated from people's actual behavior. However, hedonic pricing requires large amounts of data for robust analysis, assumes that consumers are aware of and care about the environmental variable of interest, and assumes that the market in question is not affected by outside forces. This approach can also be prone to omitted variable bias and multicollinearity. In spite of these limitations, we used the hedonic pricing method because of its ability to be applied to virtually any location.

Methods

Quantifying Ecotourism & Beach Erosion Control Co-Benefits

Our team regressed the average annual price of a standard 2-person hotel room per night on several independent variables to isolate whether and to what extent coral reefs contribute to hotel price.

Modeling Framework

An individual hotel room is considered a composite good with many different attributes contributing to the room price. These attributes include factors the hotel can influence, such as hotel characteristics and amenities offered (A), and factors the hotel has less influence over, such as environmental variables (E), location variables (L) and seasonality (S). Thus, the price (P) of an individual hotel room (i) can be modeled as a function of these variables:

Equation 1: $P_i = f_i(A_i, E_i, L_i, S_i)$

Consumers will choose hotels that offer a combination of attributes that are appealing to them and within their budget. Hedonic pricing theory assumes that the total price of a good is derived from the sum of each attribute's contribution to price (Rosen 1974). Thus, Equation 1 can be rewritten to assess the marginal contribution each attribute has to the price by means of regression analysis:

Equation 2: $P_i = \beta_0 + \beta_1 A_i + \beta_2 E_i + \beta_3 L_i + \beta_4 S_i + \mathcal{E}_i$

where β_0 is the intercept, $\beta_{1,2,3,4}$ are the various regression coefficients for each independent variable, and $\mathcal{E}_i$ is the error term. The regression coefficient of each independent variable is considered to be its marginal contribution to price. This can be equated to the economic value or willingness to pay (WTP) of a customer for a marginal change in that specific attribute, holding all other attributes constant.

Hedonic models are based on several assumptions. Firstly, the market is assumed to be perfectly competitive (Rosen 1974). This means the hotel market must be large enough with sufficient variation in attributes and consumers are aware of the range of attributes available. Consumers are assumed to maximize their utility by choosing goods with the combination of attributes most appealing to them. In hedonic regression, it is also assumed that the random error of the regression is normally distributed. Lastly, hedonic regression uses actual market prices to isolate the influence of the underlying attributes. Our study uses advertised prices instead of actual transaction prices, but the advertised prices are considered sufficiently similar to the market prices to satisfy this assumption (Rigall-I-Torrent et al. 2011).

We tested several types of regression models in the process of model specification, including linear regression, log-linear regression and log-log regression. We concluded that log-linear regression was the most appropriate for this study as it had a better fit than the other models. Better fit was assessed through diagnostic plots of the regressions, the explanatory power of the models (adjusted R^2) and the relative quality of the models (Akaike Information Criterion (AIC)). This finding aligns with other hedonic studies of environmental variables on hotel price (Rigall-I-Torrent et al. 2011, Latinopoulos 2018, Soler et al. 2019). In log-linear regression, the regression coefficients can be interpreted as a β_i x 100 % change in the dependent variable for a 1 unit change in the independent variable in question. For example, a regression coefficient of 0.45 for hotel class means a 1 star increase in hotel class is correlated with an increase in price of 45%. We accounted for seasonality by using the annual average nightly price for each standard 2-person hotel room. We ran different combinations of the following equation to assess the contributions of our variables of interest to hotel price:

Equation 3: $$ln(P_i) = \beta_0 + \beta_1 Coral\ Cover + \beta_2 Beach\ Width + \beta_3 Control\ Variables + \mathcal{E}_i$$

For our study, we are most interested in the regression coefficients on the variables of interest coral cover (km^2) and beach width (m). This data can be used to quantify the economic value that the presence of coral reefs provides directly to hotels. Coral cover measures the value of the ecotourism ecosystem service. Ecotourism includes snorkeling, diving, glass-bottom boat trips and on-reef recreational fishing activities that hotel guests take part in. The variable beach width captures the value of beach erosion control. To control for other influences on hotel price, data on hotel characteristics, several control variables for hotel characteristics, and location were included.

Hotels generate revenue by booking rooms and are interested in the preferences of their guests as an opportunity to adjust their offering to generate additional revenue. To estimate the total annual economic value to a region of an additional 1 m^2 of coral cover or 1 m of beach, the marginal willingness to pay (WTP) found in our regressions can be aggregated using the average number of hotel rooms (#Rooms) per hotel in a region, the average hotel occupancy rate (%Occupancy), the number of hotels (#Hotels), and the days in a year via the formula:

Equation 4: *Total WTP = MarginalWTP x #Rooms x %Occupancy x #Hotels x 365 days*

Equation 4 will yield the total economic value of a marginal unit of coral cover and beach width. These values can be shared with hotels to increase their awareness of the value provided to them by coral reefs.

Case Study & Data

We applied this method as a case study in 5 large tourist cities in Southeast Florida: Delray Beach, Boca Raton, Fort Lauderdale, Miami Beach and Key West. Hotel data was collected for 428 hotels in these 5 cities. After scraping tripadvisor.com for the hotel data, hotels with missing data points were removed (n = 42) , after which samples in Delray Beach (n = 10) and Boca Raton (n = 4) were deemed too small for inclusion in the study. Thus, our final sample of hotels was 372 in Fort Lauderdale, Miami Beach and Key West. Table 3 contains a complete list of variables and descriptive statistics.

Table 3: Descriptive statistics of variables used in this study.

Category	Variable	Description	Mean	Standard Deviation	Min	Max
Dependent Variable	Average Price	Standard 2-person room price ($/night) in 2018 USD	368.00	237.34	68.00	2425.00
Variables of Interest	Coral Cover	Amount of coral within buffer (km^2)	65.66	11.82	47.37	114.49
	Beach Width	Width of the beach in front of the hotel (m)	60.65	44.91	0.00	207.09
Control Variables	Distance to Reef	Distance from the hotel to the nearest reef (m)	677.71	366.98	12.25	1967.00
	Distance to Beach	Distance from the hotel to the nearest beach (m)	441.17	468.11	24.43	2125.52
	Distance to Access	Distance from the hotel to the nearest beach access point (m)	748.36	601.53	50.44	4188.35
	Distance to Airport	Distance from the hotel to the nearest airport (m)	56.30	70.19	5.53	172.88
	Dive Sites	Dummy: 1 = Reef has dive sites	0.73			
	Hotel Class	Class on scale of 1-5 stars	3.26	0.73	1.50	5.00
	Guest Rating	Hotel guest rating on scale of 1-5	3.99	0.63	1.50	5.00
	# Reviews	Number of hotel reviews	1138	1447	1	15035
	# Rooms	Number of rooms in the hotel	85.35	122.06	2.00	1504
	Airport Transport	Dummy: 1 = Hotel offers transport to the airport	0.09			
	Bar	Dummy: 1 = Hotel has a bar	0.49			
	Beachfront	Dummy: 1 = Hotel is directly on the beachfront	0.33			
	Business Center	Dummy: 1 = Hotel has a business center	0.21			
	Breakfast	Dummy: 1 = Hotel offers free breakfast	0.27			
	Concierge	Dummy: 1 = Hotel has a concierge	0.53			
	Conference Center	Dummy: 1 = Hotel has a conference center	0.17			
	Free Parking	Dummy: 1 = Hotel has free guest parking	0.21			
	Gym	Dummy: 1 = Hotel has a gym	0.30			
	Laundry Service	Dummy: 1 = Hotel has free laundry service	0.47			
	Meeting Rooms	Dummy: 1 = Hotel has meeting rooms	0.20			
	Multilingual Staff	Dummy: 1 = Hotel has multilingual staff	0.75			
	Non-Smoking	Dummy: 1 = Hotel is non-smoking	0.86			
	Wheelchair Access	Dummy: 1 = Hotel is wheelchair accessible	0.59			
	Free Wifi	Dummy: 1 = Hotel offers free wireless internet	0.86			
	City	Factor with 3 levels: Fort Lauderdale, Key West, Miami Beach				

We first applied regression analysis to the pooled data, not accounting for underlying differences in each city. However, when the control dummy variable for city was introduced, the coefficients on the variables of interest changed in significance and sign. We then re-ran local models for each city to better understand spatial variations in the effects of coral cover and beach width on hotel price. Summary statistics for each city can be found in Table 4.

Table 4. Summary of key variables for each city.

City	Variable	Mean	Standard Deviation	Min	Max
Fort Lauderdale (n = 58)	Average Price	252.65	136.94	104.00	887.00
	Coral Cover (km^2)	58.87	1.08	56.56	60.10
	Beach Width (m)	53.02	40.03	26.38	207.09
	# Rooms	93	119	2	481
Key West (n = 100)	Average Price	405.55	110.98	209.50	761.00
	Coral Cover (km^2)	83.31	6.33	74.40	114.49
	Beach Width (m)	8.21	6.70	0.00	33.32
	# Rooms	33	51	60	4
Miami Beach (n = 214)	Average Price	381.72	287.92	68.00	2425.00
	Coral Cover (km^2)	59.25	4.91	47.37	69.40
	Beach Width (m)	87.23	32.36	17.70	133.89
	# Rooms	100	140	4	1504
Overall (n = 372)	Average Price	368.00	237.34	68.00	2425.00
	Coral Cover (km^2)	65.66	11.82	47.37	114.49
	Beach Width (m)	60.65	44.91	0.00	207.09
	# Rooms	85	122	2	1504

More detailed descriptions of how we collected the data used for the study are found below.

Finding Distance to Beach & Beach Width

The team imported a shapefile into ArcGIS from the University of Florida GeoPlan Center containing the addresses of all Florida lodging facilities. For each of the five cities, we created shapefiles in ArcGIS of the start of the beach and the start of the shoreline based on the World Imagery basemap. We added points every 30 meters to each line and then used the Near tool in ArcGIS to determine the shortest distance from each hotel to the start of the beach and start of the shore. To find beach width, the distance from the hotel to the beach was subtracted from the distance from the hotel to the shore. We assume that the underlying imagery used to measure beach width was collected at the same time for all hotels in a given city.

Hotel Amenity Data

Our team used the R package rvest to scrape tripadvisor.com for hotel information. TripAdvisor was chosen due to its popularity among both domestic and international tourists for researching and booking hotels. According to Alexa, a website traffic statistics company, TripAdvisor ranks higher in domestic and global use than similar websites such as expedia.com, booking.com and hotels.com (alexa.com). The script scrapes hotel name, hotel class, guest rating, number of reviews, number of rooms and 15 other amenities for each hotel in a given region. The script also scrapes the annual range (low and high) of hotel price for a standard 2-person room per night. The average of the low and high price was used in this study. The collected data was compiled and exported in a CSV file. Using scripts ensures the reproducibility of our methods. These scripts will be made available on GitHub for other researchers to use to scrape TripAdvisor or other travel websites of choice with some adaptations.

Coral Data

The team imported a shapefile of Coral and Hard Bottom Habitats in Florida from the Florida Fish and Wildlife Conservation Commission (FFWCC) into ArcGIS, which contains the location of coral reefs off the Florida coast. To measure distance to the reef, the team created points every 30 meters along the Western reef face and measured from the hotel to the reef face using the Near tool in ArcGIS. The team created a shapefile of 50 popular dive sites along Florida's coastline using coordinates from the Florida Go Fishing website and used the Near tool to determine the distance from the hotel to the closest popular dive site. For coral cover, a 10km buffer was created around each hotel, which according to Ferrario et al. (2014) is the distance within which coral reefs provide the most direct risk reduction benefits to coastal properties. This buffer shapefile was then joined with the FFWCC coral shapefile to determine the area (km^2) of coral within each buffer.

Results

We began with a simple model accounting for the environmental variables of interest and the most important hotel variables (hotel class and guest rating). We identified hotel class and guest rating as the most important hotel variables through exploratory principal components analysis (PCA), stepwise regression analysis, and findings of similar hedonic studies on hotel price (Rigall-I-Torrent et al. 2011, Soler & Gemar 2018). We then introduced additional control variables from Table 3 one at a time to assess their effect on the coefficients of the variables of interest. The control variables were added to test the robustness of the coefficients on the variables of interest and find a specification that explained the most variation in hotel price (high adjusted R^2) while still achieving good model fit (low AIC, no multicollinearity). Table 5 shows the results from running specifications pooling all the cities together.

Table 5. Results of log-linear regression specifications for aggregated data. Standard errors are in parentheses.

Ecotourism Log-Linear Regression Results

	Dependent variable:		
	Log of Standard Hotel Room Price		
	1	2	3
Coral Cover (km^2)	0.0108***	0.0119***	0.0137***
Beach width (m)	0.0008	0.0008	0.0010**
Hotel Class	0.4519***	0.4503***	0.3801***
Guest Rating	0.0645**	0.0662**	0.0876***
Distance to Reef (m)		0.00002	0.0001
Distance to Access (m)		-0.00005	-0.0001
# Rooms			-0.0001
Beachfront			0.1070**
Conference Center			0.1903***
Constant	3.2877***	3.2332***	3.1681***
Observations	372	372	372
Adjusted R^2	0.54	0.54	0.56
Mean VIF	1.58	1.49	1.68
AIC	239.95	241.77	223.14
F Statistic	108.47***	72.6990***	
Note:		*p<0.1; **p<0.05; ***p<0.01	

Our model of choice is #3 as it accounts for the most variation in hotel price (Adjusted R^2 = 0.56) with the lowest AIC value (223.14). The specifications are not subject to multicollinearity as the mean variance inflation factors (VIF) are all low (1.58, 1.49 and 1.68 respectively). VIF is a measure of whether correlation among predictor variables is inflating the coefficients found by the model. A VIF greater than 10 is used as a threshold for multicollinearity. These specifications reveal several interesting findings for our variables of interest:

1. Coral cover is positively correlated with hotel price, however the influence is quite small. In all specifications, coral cover is significant at the 1% significance level. In model 3, a 1 km^2 increase in coral cover is expected to increase willingness to pay by 1.1%, or $3.97 for the average hotel price of $368. A 1 km^2 increase in coral cover corresponds to a 1.5% increase in coral for the average hotel in our sample.

2. Beach width is also positively correlated with hotel price, and the influence is larger than that of coral cover. This is expected because beach width likely matters more to guests than the amount of coral cover, as not all guests partake in coral-related activities. In model 3, a 1 m increase in beach width is expected to increase willingness to pay by 0.102%, or $0.37 using the

average hotel price of $368. This coefficient is only significant in model 3 and is significant at the 5% significance level. A 1 m increase in beach width corresponds to a 1.6% increase in width for hotels in our sample. The variance of beach width in our sample (2017.19) is much greater than the variance in coral cover (139.62), indicating that a wider range of beaches are being sampled.

3. Distance to reefs, distance to access points, and number of rooms were not found to be significant contributors to hotel price. Distance to access points is negatively correlated with hotel price, but not shown to be significant. Distance to reefs is unexpectedly positively correlated with hotel price, meaning that the further a reef is from the hotel, the higher the hotel price is expected to be. However, this coefficient is not significant.

Other findings include that both guest rating and hotel class are positively correlated with hotel price and are important components of willingness to pay for a hotel room. Increasing hotel class by 1 extra star is associated with a 38% increase in willingness to pay. An increase in guest rating of 1 point is associated with an 8.7% increase in willingness to pay. Both of these variables are significant at the 1% significance level. Both class and rating are considered to capture different attributes of the hotels, which may explain why many of the individual attributes were not found to have a significant impact on hotel price.

The specifications are relatively robust to the addition of control variables, except for when city is added, as in Equation 5. The introduction of the city control variable changes the significance and sign of the coefficient on coral cover. This signals that city is important for explaining variation in hotel price. For this reason, our next step was to assess the specifications in separate models for each city. A summary of the local regression results is found in Table 6.

Equation 5: $ln(P_i) = \beta_0 + \beta_1 Coral\ Cover + \beta_2 Beach\ Width + \beta_3 Control\ Variables + City + \varepsilon_i$

Table 6: Results of log-linear regression for local specifications. Fort Lauderdale is the reference level city for Model 3 + City.

Ecotourism Log-Linear Regression Results

	Dependent variable:			
	Log of Standard Hotel Room Price			
	3 + City	Fort Lauderdale	Key West	Miami Beach
Coral Cover (km^2)	-0.0032	0.0793*	-0.0008	-0.0082
Beach width (m)	0.0013**	0.0030**	-0.0019	0.0007
Hotel Class	0.3558***	0.3748***	0.1664***	0.4238***
Guest Rating	0.0813**	0.0888	0.1056	0.0406
Distance to Reef (m)	-0.00004	-0.0002	0.00003	-0.0003
Distance to Access (m)	-0.00001	0.0006	-0.00002	0.0003
# Rooms	-0.0001	0.0754	0.0013	-0.0002
Beachfront	0.1257***	-0.0789	0.1171	0.1265**
Conference Center	0.1837***		-0.0121	0.2104**
Key West	0.6209***			
Miami Beach	0.1672**			
Constant	4.1373***	-0.6930	4.9798***	
Observations	372	58	100	214
Adjusted R^2	0.60	0.69	0.32	0.57
Mean VIF	2.16	2.54	2.48	1.89
AIC	191.66	12.65	-9.09	160.94
F Statistic	51.71***	15.06***	6.17***	32.97***
Note:			*p<0.1; **p<0.05; ***p<0.01	

The city variable may be capturing spatial differences in coral among the cities or capturing the effects of variables not included in this study. The analysis by city revealed interesting trends:

Fort Lauderdale

Fort Lauderdale is the only city for which both coral cover and beach width are significantly positively correlated with hotel price. In Fort Lauderdale, a 1 km^2 increase in coral cover is associated with a 7.9% increase in willingness to pay, or $19.96 using the average hotel price of $252.65. Fort Lauderdale is the city with the lowest average coral coverage and the least variance in cover (1.16). Coral cover is relatively uniform (Figure 1). This indicates an additional unit of coral cover is more valuable to hotels in Fort Lauderdale than in the other cities. A 1 m increase in beach width is expected to increase willingness to pay by 0.3% or $0.75. Distance to beach access points is significant at the 5% significance level, meaning an increase in distance from the beach is correlated with a 0.02% lower willingness to pay.

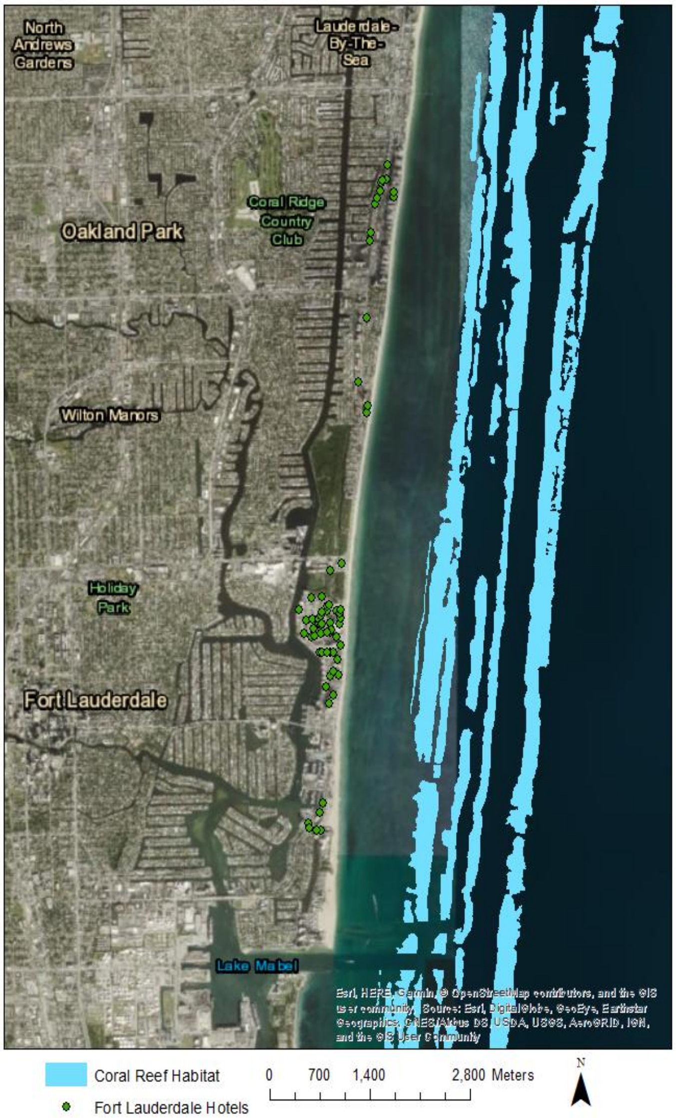

Figure 1. Location of Fort Lauderdale hotels and coral habitat.

Key West

Both coral cover and beach width are negatively correlated with hotel price in Key West, although neither are significant. Key West has the highest average coral cover with the largest variance (40.06). An additional unit of coral cover is likely not significant due to the relative abundance of coral in this city and the location of the most coral (Southeast) in relation to the concentration of hotels (Northwest), shown in Figure 2. The city has the narrowest beaches with the least variance in width (44.95). The coefficient for beach width is likely not significant due to the low variation in beach width. There is only one stretch of beach available for hotel guests to use in Key West. All other variables align with our hypothesized sign except for distance for reef, which was positively correlated, and conference center, which was negatively correlated.

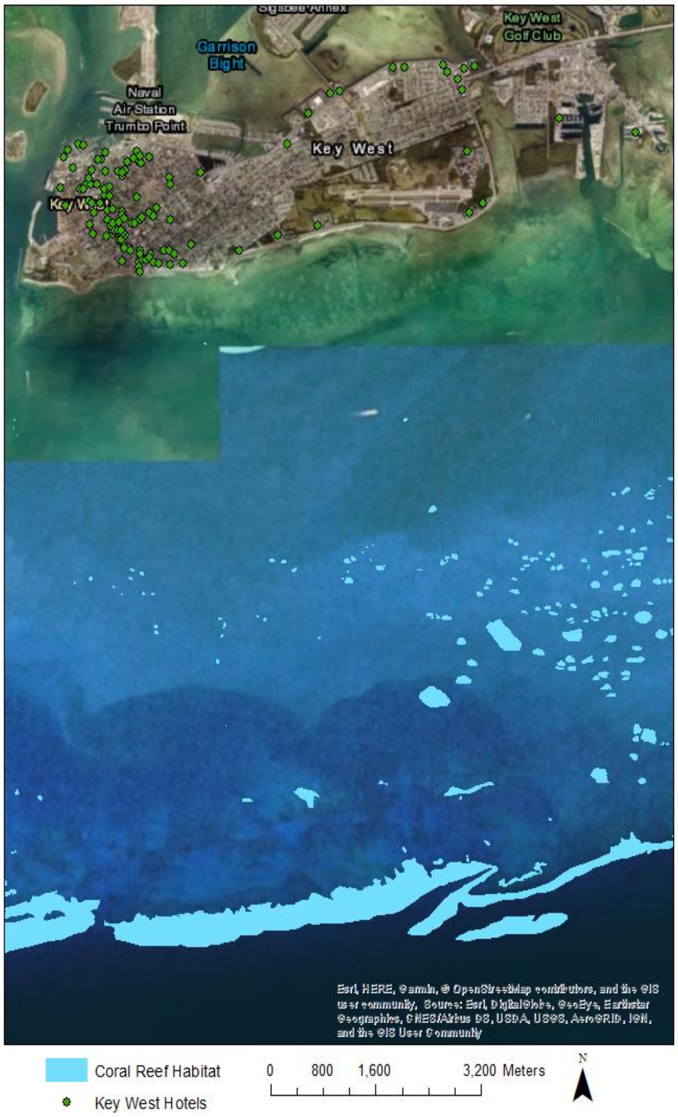

Figure 2. Location of Key West hotels and coral habitat.

Miami Beach

In Miami Beach, coral cover and beach width were not significantly correlated with hotel price. Coral cover had the least variance in Miami Beach (24.15), and beach width had the second highest variance (1047) and the widest average beaches. An additional unit of beach may not be valuable in Miami Beach due to the existing relatively wide beaches. Coral cover is likely not significant due to its low variance. All hotels, expensive and inexpensive, have about the same amount of coral cover (Figure 3), thus the regression is unable to isolate its contribution to hotel price. More control variables were significantly correlated with hotel price in Miami Beach, perhaps due to the greater sample size (n = 214). Significant variables and their association with hotel price included hotel class (positive; 42.4%), distance to the reef (negative; 0.03%), distance to a beach access point (negative; 0.03%), beachfront (positive; 12.6%) and conference center (positive; 21%). All of these variable influences aligned with our hypothesized signs. Of note is that guest rating, while found to be important in the pooled city models, was not significant in Miami Beach or any of the other local models.

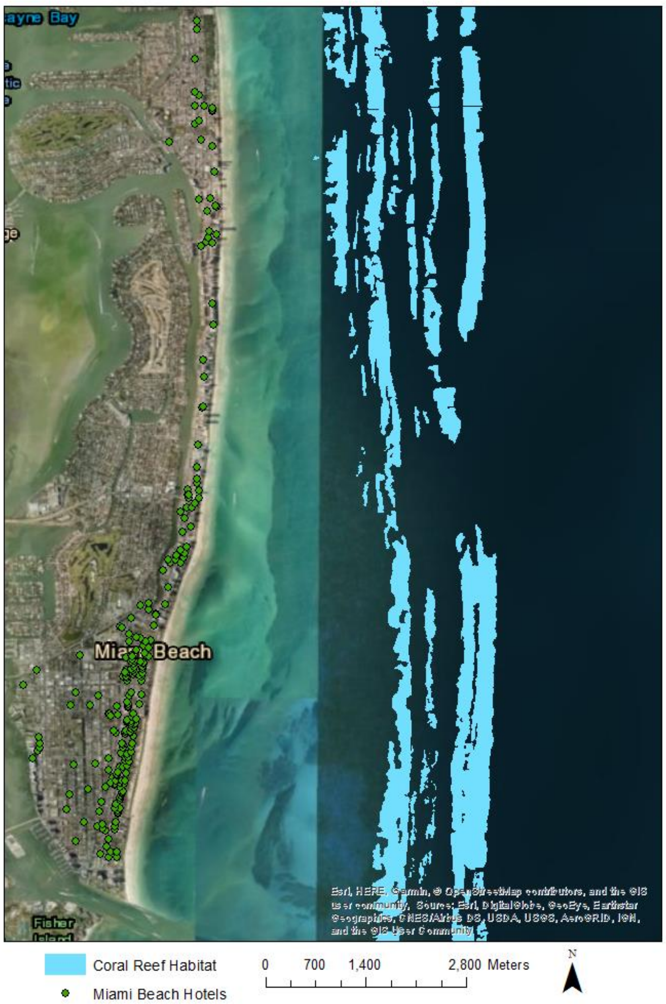

Figure 3. Location of Miami Beach hotels and coral habitat.

Quantifying Total Economic Value

As outlined in the methods section, Equation 4 can be used to quantify the total annual economic value that coral cover and beach width contribute to hotels. Applying Equation 4 to Fort Lauderdale using the 2018 annual average occupancy rate of 86.9% for Broward County (STR 2018) and data from Table 4, we find that:

Total WTP_{coral} = *\$19.96 x 93 x 86.9% x 58 x 365* = ***\$34.15 million per km² of coral cover***

Total $WTP_{beachwidth}$ = *\$0.75 x 92.69 x 86.9% x 58 x 365* = ***\$1.28 million per m of beach width***

Restoration costs for coral projects are usually quoted in per m^2 units. We can convert the $35.14 million per km^2 of coral cover to $35.14 per m^2 of coral cover for a direct comparison. This aggregation clearly indicates that guests value the beach erosion control services provided by coral reefs more than the ecotourism services. This was expected as only a subset of hotel guests are expected to participate in ecotourism activities, while a larger subset is expected to visit the beach.

Discussion

Our analysis applied the hedonic pricing method to nightly standard hotel room price to isolate the value coral reefs provide directly to hotels via the ecosystem services of ecotourism and erosion control. Our results indicate that both coral cover and beach width are significantly positively correlated with hotel price, and that this correlation depends on the specific region. These findings have implications for both private and public stakeholders, and especially hotels.

The economic value found with our method can be added to the protection value provided by reefs to find a total economic value of reefs to hotels. TNC has quantified the protection value and assessed the cost-effectiveness of reefs and other nature-based coastal defenses (Beck et al. 2018, Reguero et al. 2018). However, TNC's previous work does not include the additional ecosystem services included in our study. Our methodology fills this gap in knowledge and enables TNC to provide more accurate cost-benefit analyses to coastal stakeholders and potential investors.

Our findings align with other studies that show coral reefs are valuable for the tourism industry. For example, Spalding et al. (2014) found the highest value coral reefs in Florida provide an additional $908,000 to the tourism industry annually. In contrast, our finding valued willingness to pay of hotel guests in Fort Lauderdale for an additional km^2 of coral to be $34 million. Both studies find a significant contribution, but differ greatly in magnitude and experimental design. These studies indicate that coral reefs are important for tourism and hotel revenue. A WRI report found that coral reef degradation could lead to annual economic losses of $350-870 million in the Caribbean region (Burke et al. 2004). Coral reefs are important to protect for economic prosperity in this region. Valuation methods such as ours play an important role in attracting new investment in coral reefs.

An interesting finding of our study is that coral cover and beach width are only significant and positively correlated with hotel price in Fort Lauderdale. We hypothesize this is because Fort Lauderdale is more dependent on reef tourism than Miami Beach and Key West. This indicates that location plays a role in the value of coral reefs. Both Miami Beach and Key West are larger cities that receive more annual tourists than Fort Lauderdale (Visit Florida 2019). Our data collection also found the most dive sites in Fort Lauderdale (61). This indicates that our method is more appropriate to apply in areas that are more reliant on reef tourism, such as smaller Caribbean countries.

Limitations & Opportunities for Future Analysis

The hedonic pricing method is an effective way to determine how coral reefs influence hotel prices; however, there are some limitations to our model. First, we assume that hotel guests make informed choices when booking a room. Although hotels may advertise activities such as snorkeling and diving on their websites or on TripAdvisor, customers may not be aware of these activities when booking a hotel. This means the hedonic assumption that customers are fully informed may not be completely satisfied. Second, we only used current hotel and coral cover data in our regression analysis. Ideally, we would run a panel data regression, which compares current hotel and coral data to historical values. However, this data is difficult to access, so we were unable to incorporate this analysis into our project.

While our regression model shows a positive correlation between hotel price and beach width and coral cover in Southeast Florida, this region may not be the most appropriate to apply as a case study. The three cities we used in our model practice beach nourishment, which likely impacted our results by over- or underestimating the importance of corals to beach preservation. As a future opportunity, our model could be applied to small countries in the Caribbean that do not nourish their beaches. Tourism accounts for 15% of the Caribbean's gross domestic product (Wielgus et al. 2010). Due to the importance of tourism to the Caribbean, our regression model may indicate that coral cover and beach width more strongly predict hotel price in this region.

Our analysis relied solely on data collected from tripadvisor.com, which may have influenced its quality. Many of the amenities in TripAdvisor are manually entered by the hotels, and may not be accurate. For example, during data collection, we noticed some hotels did not advertise the Pool amenity, but had pools in pictures posted by users. Furthermore, TripAdvisor states the annual range (low and high) for nightly price on the hotel overview page is for a standard 2-person room. However, it is unclear how this is determined for each hotel. Also, taking an average of the low and high price may over- or underestimate actual annual average nightly price, as the extreme prices likely happen infrequently during the year. This analysis could be validated by using prices for specific days and compared to the values found in our study.

Objective 2. Identifying Costs of Reef Restoration

Introduction

To secure funding for a reef restoration project, an estimate of restoration cost must be broken down into a realistic budget that can be given to investors. As reef restoration is a relatively new field, there is a large gap in information within the literature surrounding restoration cost. Most of the documented restoration projects do not report their costs or fail to outline what is included in their cost estimates. A recent study by Bayraktarov et al. (2016) examined 954 restoration projects, and noted that of these observed projects, only 33% reported any cost data and 28% described what these costs included.

The lack of existing data in the literature shows a need for a more structured and concrete approach to determining restoration cost. We contacted professionals in the reef restoration field to gather their expertise on why cost information is so underreported within the literature. This allowed us to also collect cost data on the projects that our interviewees were currently working on. These interviews later led to the development of an Excel model that will calculate and report a total cost for restoration by adding the many stages and components of restoration projects.

Methods

Literature Review: Coral Restoration Costs

An extensive review of the literature revealed that costs of coral restoration projects are extremely complex and vary greatly by location and technique. An accurate cost assessment of coral restoration must consist of all resource inputs, including: 1) capital costs, i.e. pre-construction and construction costs, 2) operational costs, i.e. maintenance and monitoring costs and plan management, and 3) labor force costs, i.e. supervision, training, and manual labor costs (Spurgeon and Lindahl 2000). A Ferrario et al. (2014) study found a median project cost of US$1,290 per square meter (2012 dollars). BioRock structures implemented in various countries were the least-costly restoration method according to the literature, estimated at US$129 per square meter (Ferrario et al. 2014). Concrete structures with coral transplants located in Florida were the most expensive projects, estimated at US$927 per square meter (Goreau and Hilbertz 2005). Furthermore, coral restoration projects in low-income/middle income countries are half as expensive as those in wealthier countries, mostly due to the reduced cost of labor and materials. According to a study by Bayraktarov et al. (2016), the average total restoration cost (accounting for capital and operating costs) for 8 sites located in high-income countries is US$550 per square meter (2010 dollars), while the average total restoration cost for 8 sites located in lower-income countries is US$33 per square meter. According to these studies, coral restoration projects located in the Caribbean will likely cost an order of magnitude less than those located in the United States due to variations in cost of labor and materials.

Literature Review: Grey Infrastructure

To further examine the options that hotels and other coastal property owners have in protecting their infrastructure, we considered the alternative to reef restoration for coastal protection: grey or artificial infrastructure, such as seawalls and submerged breakwaters. We assessed the cost of both coastal defense scenarios so that The Nature Conservancy can present a comparison to potential investors, further educating them on the key cost differences between artificial and natural infrastructure. This information is useful for hotels and decision makers when considering where coastal protection funds should be allocated.

The lifetime of most artificial shoreline protection structures is generally less than 50 years, and the increase in sea level and storm intensity will require more maintenance and reconstruction of intact structures over time (Nordstrom 2014). Estimates show that the global cost of protecting coastlines with grey infrastructure, such as dikes and levees, will be upwards of $12-71 billion in the year 2100 (Hinkel et al. 2014). On average, the construction of reef restoration projects is significantly cheaper than the costs of building artificial breakwaters (Ferrario et al. 2014). While grey infrastructure may prove costly, a hybrid of grey and natural infrastructure could provide the most cost-effective solutions to coastal protection, especially around grey infrastructure that is already in place (Monks 2017). Monks (2017) suggests that in many cases, a combination of grey and natural infrastructure would be the optimal solution for coastal risk reduction. Artificial structures are able to provide immediate wave attenuation benefits, while coral reefs take years to grow to their full protective potential. Corals can be planted on this artificial substrate, further attenuating waves and providing additional ecological services as they grow (Monks 2017).

Additionally, there is evidence that grey infrastructure can have adverse effects on coastal habitats. These structures do not function as a supplement for natural rocky habitats, and the introduction of grey infrastructure can cause fragmentation and even loss of habitat along coastal regions (Bulleri et al. 2010). These structures are also costly to remove, and many communities leave them to deteriorate over time, further harming natural habitats and creating ongoing management costs for coastal communities (McQuarrie et al. 1998). Further analysis is needed to determine how artificial infrastructure affects marine habitats, and whether investments in natural infrastructure would be more cost effective than their artificial alternatives (Sutton-Grier et al. 2015).

While the cost of grey infrastructure is highly variable and depends on the size of the project, the shape of the beach, and the water depth at each site, it can be averaged in specific areas to find the cost per linear foot or linear meter. For the purpose of this project, we have found the median cost from several different studies. An example study by the USGS in 2014 found that the median cost for building artificial breakwaters was just below $20 thousand per linear meter, which is substantially higher than the estimated $1.3 thousand per m^2 that it costs for coral reef restoration (Ferrario et al. 2014).

Expert Interviews on Coral Reef Restoration

Since our literature review revealed that the cost of restoration is highly variable and depends on many factors that will change with both location and restoration technique, we decided to contact professionals in the field. Over the summer and fall of 2018, we conducted telephone interviews with restoration professionals, mainly in Florida, to discuss the overall costs of projects that they were currently working on. We contacted 18 professionals within 11 different organizations, including NOAA, The World Bank, and the Coral Restoration Foundation. The questions that we asked our interviewees can be found in Figure 4 in the Appendix. We focused our interviews on restoration professionals in Florida since that is where we applied our benefit valuation method. There was useful qualitative data that came from these interviews, although most of the interviewees were unable to provide concrete cost numbers, either because they did not have access to all of the data or because the numbers were so variable per site that many had never tried to total them before. While those that we interviewed seemed enthusiastic about our project and were eager to help, they mentioned that they would not be able to provide a complete cost breakdown for their projects due to the complexity and many stages of reef restoration.

The interviewees were able to provide us with a better understanding of all the different components that go into a restoration project. These interviews showed us how restoration cost can vary based on location, organization, and technique. Costs of reef restoration can include project preparation (nursery care, project engineering, and permitting), reef materials (hard costs), labor (outplanting installation, project management, staff training, etc.), and post-implementation costs (monitoring and maintenance costs for X number of years). Project cost also depends on the reef parameters, such as the height, width, density, and depth of each restoration project. The use of volunteers was another important variable in determining cost for a project. If volunteers could be retained for several weeks/projects, then an organization could save money with their cost of labor. If volunteers only helped for a single trip/project and if training new volunteers was a recurring task, then the use of volunteers could prove to be too time-consuming and costly.

Results of Expert Interviews

From the cost data we received from the expert interviews, we have created a rough estimate of $60-90 per m^2 for coral restoration cost (for Miami, Florida, where the majority of interviewees currently work). These numbers are substantially lower than the Bayraktarov (2016) study for restoration costs in the US, likely because their study looked at restoration projects from up to 20 years ago, and the cost of restoration has drastically decreased over the years. As nursery projects scale up in size and continue to grow hardier species of coral, the cost per coral fingerling decreases. While the interviewees were able to provide some numbers, they expressed a lack of confidence in the accuracy and completeness of their cost estimates. They mentioned that they excluded certain cost factors and key steps along the restoration process that they did not have access to, and reminded us that each project would be highly variable.

None of our interviewees could provide an average cost of restoration, as cost will always vary by location, organization, and growing and planting technique. The $60-90 per m^2 is an estimated range based on the numbers that were provided to us, but can now be compared to the estimates of grey infrastructure cost from the same region. We contacted the Army Corps of Engineers, who are responsible for funding and constructing some of the larger grey infrastructure projects along the US coastline. The US Army Corps of Engineers in Jacksonville, Florida reported an estimate of $6,234 per linear meter of vinyl sheet pile and $7,874 per linear meter of steel sheet pile. These numbers only include the cost of raw materials and do not account for labor costs. Still, both our interviews and literature review of cost estimates revealed that the cost of grey infrastructure is substantially higher than the cost of reef restoration. Based on our interviews, we also observed that it is difficult for restoration groups to determine and report their costs of restoration. In order to facilitate the process of gathering and reporting costs, we created a model that can be used by organizations to plan future restoration projects.

Restoration Cost Model

We have worked with our clients at The Nature Conservancy to construct an Excel model that determines an overall cost of restoration by collecting and aggregating all of the smaller components that go into restoration cost (see Appendix, Figure 5). The elements in the model have been separated into three phases: the Preparation Phase, the Implementation Phase, and the Post-Implementation Phase. Separating the cost into three phases will make it easier to track the different variables that contribute to cost, and will allow both restoration organizations and investors to better visualize the many factors involved. Each cost variable within the model has been mentioned in the literature (Spurgeon and Lindahl 2000) and later confirmed through our expert interviews, and plays an important role in the overall cost of reef restoration. With this model, even if firms and organizations are unsure about the overall cost of a project, they can include individual components (i.e. inputs in blue on the left side of the spreadsheet - Appendix, Figure 5) or comment on the individual phases (preparation, implementation, and post-implementation). The current form of the model is an Excel spreadsheet that requires cost inputs and delivers total outputs per each phase. Organizations can share their spreadsheet with other groups or consultants who are helping with a given project. This way all of the cost info is in a single document and can be easily accessed.

The model will provide restoration groups with a resource that can aggregate their costs of restoration at different sites based on actual numbers or cost percentages, while making sure to include all key steps along the process. The model has been sent to the restoration professionals that we interviewed over the summer and refined several times based on the feedback they provided. Our model differs from spreadsheets that restoration groups currently use to evaluate their costs. One major difference is that our model includes the nursery costs and outplanting costs together in the same spreadsheet. Keeping these stages together makes the overall cost estimate more complete. Our model also includes the permitting costs, diving gear, and other artificial (hard) costs that a project may have.

Discussion

There is a large gap in data that exists within the literature relating to the cost of reef restoration. As this is a relatively new field, many project costs have gone unreported, especially in countries outside of the US. Our literature review showed us that grey infrastructure, such as submerged breakwaters, is generally more expensive than coral restoration (Ferrario et al. 2014). The costs of both grey infrastructure and reef restoration are highly variable, as there are many components that contribute to project cost, and each of these can fluctuate based on location and technique. Therefore, we did not find a universal average cost of reef restoration. While the interviewees over the summer provided their expertise on the components of reef restoration, they were unable to provide us with concrete cost data.

Our interviews with experts in the field left us with several main takeaways. They all mentioned how restoration is becoming cheaper as projects are scaling up, and that restoration is becoming more streamlined and deliberate. As nurseries are growing in size, the cost per coral decreases. We also learned that labor is the main component of cost for restoration projects, and finding ways to lower labor costs can have major impacts on a project. This can include effective use of volunteers or low-paid interns and avoiding the use of high-paid scientists for the outplanting of coral. We also learned that none of our interviewees were growing coral for the purpose of coastal resilience, and that each project had the sole objective of habitat creation for conservation and tourism purposes. Often projects will be funded through private interest groups, such as the Florida Tourist Development Council or the Ocean Reef Club, who fund restoration so that tourists can continue to recreate among healthy reefs. By making the connection between coral restoration and coastal resilience, The Nature Conservancy will inspire reef restoration for the purpose of coastal protection and not just for habitat restoration purposes. This could motivate new sources of funding to support more substantial restoration projects.

Based on the information we gathered through these interviews, we developed a model that can estimate the overall cost of a restoration project by summing the many smaller components of reef restoration. Our cost model will allow organizations to construct an accurate total restoration cost without leaving out key components, share cost information with other restoration groups, and provide potential investors with a more complete budget for a given project.

Limitations & Opportunities for Future Analysis

Our cost model provides an opportunity for restoration groups to more completely and accurately report restoration costs. Our model will allow groups to organize and deliver their costs to secure financial backing from grants or public funding. The current model does not include the density at which coral is planted, the survivability of coral after it is planted, local oceanographic conditions, or whether the nursery is on land (ex situ) or in the ocean (in situ). A more complete model should include these additional variables, as each impacts project cost.

The next phase of this model could include a user-friendly interface where organizations are given the option to select which restoration techniques and materials they are using, and a total cost of restoration would be generated based on their responses. A future version of this model could be created on a software program that asks the user questions about their restoration project and constructs a budget based on an algorithm that incorporates the many stages of reef restoration, taking into account the timeline and any barriers that a project might face.

Objective 3. Identifying Future Suitable Habitat for Coral Restoration

Introduction

In addition to evaluating the co-benefits and costs of restoration, it is important to evaluate the feasibility of successful projects by considering where corals are today and where they can exist in the future. Corals inhabit specific waters along the coasts of continents and islands based on key environmental conditions that dictate settlement, growth, and survival. With growth rates of 0.3 to 2 centimeters per year for large corals and up to 10 centimeters per year for branching corals, it can take up to 10,000 years for a coral reef to form from a group of larvae (Barnes 1987). Restoration can aid this process by maintaining coral populations in the event of damages; however, feasibility assessments should consider how climate change will impact the ability for corals to survive in regions where they currently provide risk reduction benefits, in addition to the co-benefits specific to the region. To do this, we considered the following -

1. Which Caribbean reef-building coral species characterize the large reef ecosystems that provide a risk reduction benefit,
2. What oceanographic conditions are most relevant for determining current coral reef habitat, and
3. How coral suitable habitat will change in future oceanographic conditions.

Species of Interest

In the greater-Caribbean region, the physical structure of coral reefs are composed of reef-building taxa such as those found in the families *Acroporidae, Montastraeidae, Merulinidae*, and *Poritidae*. However, certain key members of these families, such as *Acropora palmata* (elkhorn coral), *Acropora cervicornis* (staghorn coral), and *Montastraea annularis* (boulder star coral) are critically endangered due to climate change and anthropogenic effects (Aronson et al. 2008a, Aronson et al. 2008b, Aronson et al. 2008). As reef-building corals, these species are widely spread throughout the Caribbean and often serve as the foundation for other coral species (Putnam et al. 2017). These same corals are often the attraction of ecotourism along coastal cities and provide wave height reduction benefits to coastlines (Spalding et al. 2014). For this analysis, four coral families (*Acroporidae, Montastraeidae, Merulinidae,* and *Poritidae*) were chosen as a result of their important risk-reduction capacity, as well as ecotourism and beach erosion reduction co-benefits.

Oceanographic Conditions

Coral formation depends on dozens of environmental conditions. Previous studies predicting habitat suitability for coral reefs on a global scale performed correlation tests to identify the key variables that determine suitable habitat for coral reefs (Couce et al. 2012; Freeman et al. 2013). While differing between one or two variables, these studies suggest key variables are $CaCO_3$ saturation state of seawater, sea surface temperature, salinity, photosynthetically active radiation, current velocities, and nutrients such as phosphate and nitrate (Couce et al. 2012; Freeman et al. 2013; Silva and MacDonald 2017). Each significant variable has a specific impact on corals, ranging from coral bleaching to decreased calcification as a result of increased acidity in the ocean (Table 7). Detail on each individual variable is outlined below.

Table 7. Impact of six key variables on coral reefs.

Variable	Impact	Citation
pH	Decreased calcification and coral bleaching	Kleypas and Yates (2009)
Sea Surface Temperature (SST)	Coral bleaching	McClanahan et al. (2007)
Salinity	Decreased rate of photosynthesis	Ferrier-Pages et al. (1999)
Photosynthetically Active Radiation (PAR)	Decreased calcification and coral bleaching	Allemand et al. (2011); Marsh and Svensmark (2000)
Current Speed	Magnify negative effects of other variables	Comeau et al. (2014)
Nutrients	Coral bleaching and increased phytoplankton and algae concentration, which can lead to competition for space	D'Angelo and Wiedenmann (2014)

<u>pH</u>

Fossil records suggest that corals are most vulnerable when rapid increases in sea surface temperature and CO_2 occur, due to the resulting decline in $CaCO_3$ saturation (Pandolfi et al. 2011). Corals thrive in stable environmental conditions, and the process by which the ocean absorbs CO_2 causes the ocean's pH to decrease. A lower pH catalyzes the conversion of carbonate ions (CO_{3-}) to bicarbonate ions (HCO_{3-}), which in turn decreases the $CaCO_3$ saturation state of seawater and limits coral reef calcification rates (Lunden et al. 2013). Corals are found to thrive when pH is between ~7.8-8.1 (Fabricius et al. 2011). With this in mind, we decided to use pH as one of our indicator variables to capture the effects of a changing $CaCO_3$ saturation state of seawater.

Sea Surface Temperature (SST)

The optimal range of SST for coral growth is 73-84°F (Freeman et al. 2013). Thermal stress from an increasing SST leads to bleaching, a process which results in corals expelling their photosynthetically-active zooxanthellae. Zooxanthellae are vital for coral survival as they are the energy-producing component of reefs. Their loss results in the slow starvation and eventual death of coral species. Along with this, the zooxanthellae are what give corals their vibrant colors; a loss in these microorganisms is a direct loss of ecotourism attraction for diving and snorkeling. Additionally, long term coral bleaching disrupts the structural integrity of reefs (Graham et al. 2008). This loss in structural integrity in turn results in a decrease of the risk reduction benefit corals provide.

Salinity

Sea surface temperature is closely tied to seawater salinity (Durack and Wijffels 2010). As global temperatures increase, salinity also increases as a result of the interaction between evaporation, wind, rainfall, and the melting and freezing of sea ice. Certain Caribbean coral species, such as Acroporids, thrive in normal marine salinities such as 33-37%, and most reef-building corals are intolerant to salinities less than 25 parts per thousand (NOAA National Marine Fisheries Service; Freeman et al. 2013). Salinity has a direct influence on coral biology as it influences protein concentration and rates of photosynthesis, with the ideal conditions at lower concentrations (Ferrier-Pages et al. 1999).

Photosynthetically Active Radiation (PAR)

The zooxanthellae within corals require visible light (photosynthetically active radiation, PAR) to provide nutrition to its coral host as a result of photosynthesis (Hoegh-Guldberg 1999). Coral reefs are found in lower latitudes where PAR levels are at their maximum, and at specific shallow depths where the ocean allows for radiation to penetrate (Marsh and Svensmark 2000).

Current Speed

Ocean current velocity is immensely important to coral life history and suitable habitat. Currents not only direct coral larval transport to downstream reefs, but they also affect oxygenation, nutrients, and heat stress (Riegal and Piller 2003; Woesik 2001; McClanahan et al. 2005). Ocean warming is expected to impact current speeds by shifting thermocline depths resulting in effects on mixing within the water column (Steinberg 2007).

Nutrients

Studies have found that phosphate and nitrate are important for coral suitable habitat, as an increase in these nutrients results in increased algal growth, which outcompete corals for nutrients and affect availability of PAR (McManus and Polsenber 2004). However, our initial modelling found that nutrient concentrations were homogeneous throughout the study area and did not contribute to the model. Furthermore, Couce et al. (2013) found that nutrients were not most relevant for coral species distribution modelling through a boosted regression trees (BRT) analysis. BRT is another species distribution modelling technique which improves a single model by fitting many models and combining them to predict habitat (Elith et al. 2008). Given these results, we excluded this variable from our analysis.

Future Conditions

Representative Concentration Pathways (RCPs) are greenhouse gas concentration and emissions trajectories designed to support research on impacts and policy responses to climate change. Two climate scenarios were chosen for this analysis: “RCP 8.5” and “RCP 4.5.”

RCP 8.5 is our current ‘business as usual’ trajectory of climate change, which corresponds to a high greenhouse gas emissions pathway leading to a radiative forcing of 8.5 W/m^2 by the year 3000. This climate trajectory assumes the following: high population, relatively slow income growth, and modest advancements in technology and energy efficiency (Riahi et al. 2011). These assumptions result in a long-term high energy demand with little to no climate change policy to curb greenhouse gas (GHG) emissions.

RCP 4.5 is a scenario reflecting a stabilization of radiative forcing to 4.5 W/m^2 by the year 2100. This scenario involves a reduction and stabilization of radiation achieved through climate policies, specifically tied to energy production and land use, that limit GHG emissions (Thomson et al. 2011). To develop RCP 4.5, a global change assessment model (GCAM) is used as a reference scenario. This GCAM scenario depicts a world with a global population maximum of over 9 billion in 2065 and a decrease to 8.7 billion in 2100, as well as a large increase in global GDP, and a tripling of global primary energy consumption (Clarke et al. 2007). RCP 4.5 uses the GCAM as a reference and applies GHG emission policies specifically targeted at the reduction of CO_2, CH_4, N_2O, HFCs, PFCs, and SF_6, and chemically-active gases such as carbon monoxide (CO) and volatile organic compounds (VOCs) (Thomas et al 2015). These reductions are achieved through prioritizing 1) global primary energy consumption on sustainable fuel sources, 2) global electricity production on sustainable sources through technological advancements, and 3) efficient changes in global land use and cover such as increasing green space and intercropping.

These climate scenarios, and associated GHG emission levels and external forcings, can be incorporated into climate models which trace the long-term trajectory of effects these forcings and emissions have on complex global processes. The Community Earth System Model (CESM1) Large Ensemble Project is a climate model which couples atmosphere, ocean, land, and sea ice components (Kay et al 2015). For this specific analysis, the CESM1 produces diagnostic biogeochemistry calculations, including our variables of interest for the two climate scenarios through the year 2080.

For these five variables under RCP 8.5, we expect an overall increase in salinity, SST, and PAR, and a decrease in pH and current speeds. We see a similar trend under RCP 4.5; however the shift is proportionally lower (see Appendix, Table 8). How these changes affect coral suitability is discussed further below.

Methods

Maxent

Our goal is to estimate the range of the four reef-building coral families throughout the Caribbean region. In order to predict future suitable habitat for coral reefs, various species distribution modeling (SDM) techniques were considered. The probability of coral presence was modeled using a SDM technique called maximum entropy modeling (Phillips et al. 2006; Phillips et al. 2009). The program Maxent relates species location records with specific environmental or spatial characteristics across the landscape to conduct 'niche modeling,' a technique that has been applied to a wide array of fields of ecological and environmental study (Franklin 2009). SDMs are becoming frequently used for ecological and conservation purposes as species location records are increasing in availability and the ability to apply these methods is now available within multiple interfaces, such as RStudio and ArcGIS (Gomes 2018).

Maxent uses a combination of species presence data with a variety of spatially explicit environmental layers (climatic, biogeographical, etc.) to build species 'niches,' or 'bioclimatic envelopes,' i.e. environmental profiles of high habitat suitability. It applies an algorithm that quantifies the unknown probability distribution for a species across a user-defined landscape without inferring any information about the observed distribution, creating a least biased prediction (Slater and Michael 2012; Merow et al. 2013). Maxent then produces a probability distribution of maximum entropy (i.e. that which is closest to uniform), subject to constraints imposed by these layers (Philips et al. 2006; Slater and Michael 2012). Once a profile for the species of interest is determined, this profile can be used to predict species ranges beyond the limits of known localities to a certain amount of statistical certainty. The accuracy of these range predictions relies heavily on the input parameters, understanding of basic ecology of the species, and the program settings used when running the program (Merow et al. 2013).

Some limitations to consider for Maxent are the following:

1. Accuracy and availability of historical and current presence points for species of interest
2. Consideration of only species presence points, as opposed to species characteristics (e.g. health)

Despite these limitations, range outputs from Maxent give researchers a clear tool to conduct future surveys, identify regions of conservation concern, or predict changes in species ranges under different climate or biogeographical conditions. Given Maxent's high predictive performance when compared to other modeling techniques, this technique is ideal for our analysis (Elith et al. 2009).

Maxent Model Parameters

Our Maxent model runs used multiple settings following the Merow et al. (2013) guide for modeling species distributions using Maxent. Once intended constraints and settings are decided upon, the mathematical algorithm is adjusted according to these settings. First, we applied the default setting of 10,000 background points, or absences, to the model. Maxent contrasts these background points, where species presence is unknown, against the known species presence locations (Merow et al. 2013). This is an important characteristic of Maxent because background points are used to predict presence probabilities; Maxent initially assumes the species of interest is likely to be present anywhere in the user-defined landscape, and modifying the background sample size will adjust how the species localities are assumed to be distributed (Merow et al. 2013). As the four coral families are found relatively widespread along the coasts of Caribbean countries, we assumed default settings for background points is sufficient to predict presence probabilities. Additionally, species presence predictions beyond their habitat range were removed via a depth mask following the model run.

Next, default settings for feature classes were turned off in order to remove the threshold and hinge features from model runs, while leaving linear, quadratic, and product feature settings on. Threshold features create a binary predictor sequence in which values above the threshold receive a 1, while those below are 0; hinge features act similarly, however, apply a linear function (Merow et al. 2013). When hinge and threshold features were first applied, response curves for each environmental layer returned no discernable pattern suggesting there may be too much flexibility in our settings. As a result, only the linear, quadratic, and product features were applied to each model run to maintain structure and minimize noise in the model. According to Merow et al. (2013), the linear feature verifies that the mean values of the environmental layers in locations where the species is predicted to occur matches the mean values where they are observed, while the quadratic feature constrains the range of each environmental layer at each predicted presence point to match observations. Furthermore, the product feature constrains covariance between each environmental layer, acting similarly as interaction terms in regressions.

Lastly, our Maxent runs applied a jackknife test of variable importance. In a jackknife procedure, each variable is individually removed from a model run in order to determine how each variable contributes to the AUC (Slater and Michael 2012). The AUC, or area-under-curve, is a model fit metric with a maximum value of 1.0 and can be interpreted as the probability that a randomly-chosen presence point is ranked higher than a randomly-chosen background point (Merow et al. 2013). In other words, the greater the AUC, the greater the model fit. This is an important feature to include in order to effectively analyze Maxent outputs and improve future runs. While identifying and minimizing covariance prior to Maxent analyses is highly recommended via a principal components analysis, a jackknife test can identify covariance and confirm appropriate use of input variables (Freeman et al. 2013; Merow et al. 2013). Previous studies performed a principal components analysis to identify the most relevant and important environmental variables to predict suitable habitat for coral reefs, as well as a jackknife analysis which further verified appropriate use of each variable (Freeman et al. 2013; Couce et al. 2013). As a result, we chose to include the same input variables (e.g. SST, salinity, pH, PAR, and current speeds) in our analysis.

Maxent Input Variables

I. Coral Occurrence Data

Caribbean *Acroporidae, Montastraeidae, Merulinidae,* and *Poritidae* species occurrence data were taken from the Global Biodiversity Information Facility (GBIF), a database of georeferenced locality data from museum records, herbariums, and other open access research institutions. Downloaded using RStudio, this file was clipped to an extent containing the greater-Caribbean region, and outliers and duplicate observations were removed to create a dataset containing only species names and associated latitude and longitude. This dataset contains ~ 2,200 records of *Acroporidae, Montastraeidae, Merulinidae,* and *Poritidae* species, such as *A. cervicornis*, *A. palmata,* and *M. annularis*. Species observations for the four families were analyzed by Maxent in one dataset and therefore were not distinguished by family. Maxent analyses were initially run separating each family; however, each output consisted of nearly identical prediction maps because these species coexist throughout the Caribbean region.

II. Environmental Layers

To model current and future suitable habitat for coral reefs, the mean values for the five environmental layers (pH, SST, salinity, PAR, and current velocities) were taken from the

Parallel Ocean Program (POP) Ocean Model as part of the Community Earth System Model (CESM1) Large Ensemble Project (Kay et al 2015). The CESM1 is an open-access, readily available climate model including the atmosphere, land surface, ocean, and sea ice.

The POP Ocean Model produces a variety of oceanographic environmental variables at an ocean depth of up to 5,375 meters covering two time periods: 1920 - 2006 and 2006 - 2080. Multiple netCDF datasets were used to model present (2008 - 2018), near-future (2040 - 2050) and future (2070 - 2080) suitable habitat for coral reefs at a depth of 35 meters based on the reef-building corals that offer protection for tropical coastlines.

Using RStudio, the raw 1° resolution global variables were averaged across our desired time periods (2008 - 2018, 2040-2050, 2070-2080) and cropped to the Caribbean extent. Next, using ArcGIS, the present, near-future, and future raster datasets were defined to the GCS WGS 1984 projection and interpolated to 0.083° resolution across adjacent waters to fill data gaps. Differences between the (1) near-future and present and (2) future and present CESM modelled conditions were then applied to present observed values derived from the Bio-Oracle Marine Data Layers and Global Marine Environment Datasets (GMED) and converted to ASCII files for use in Maxent. These datasets provide observed values of our variables of interest as the basis for the statistical downscaling and data gap interpolation of the CESM output models in ArcGIS (Basher et al. 2018). Variables from Bio-Oracle include pH, SST, salinity, and PAR, while ocean current speeds were taken from GMED. Bio-Oracle provides data on 18 variables and GMED provides data on 48 variables on present day environmental conditions in 0.083° resolution in the GCS WGS 1984 standard projection.

Finally, after each Maxent model run, ArcGIS was used to display the five scenarios of current and future habitat suitability for all four reef-building coral families. In order to consider the effect ocean depth has on coral suitability, depths >100m were masked from each Maxent output (Freeman et al 2013; Basher et al. 2018). Lastly, difference maps were created by subtracting the masked present suitability map from the masked future suitability maps.

Results

Compared to the present coral suitability map (2008 - 2018) predicted by Maxent, there is an overall loss in suitable habitat across the Caribbean region under RCP 4.5 and RCP 8.5 scenarios in near-future (2040 - 2050) and future (2070 - 2080) periods. By 2080, results indicate a greater loss of suitable habitat in the RCP 8.5 scenario compared to RCP 4.5 with a larger proportion of areas expected to undergo a decrease, as opposed to a stabilization or increase in suitable habitat (see Appendix, Figure 8, 10).

In the present Maxent prediction (2008 - 2018), there is notable overlap of species presence points and areas of high suitability, indicating observed and modelled coral reef presence are in agreement (AUC = 0.722) (see Appendix, Table 9; Figure 6). While models with an AUC > 0.75 are considered a useful prediction, the AUC for all suitability maps may have increased because depths >100m were masked following each Maxent model run (Phillips and Dudík 2008). Geographic masks can be applied prior to or following a Maxent analysis. A previous study found that a geographic mask applied to each environmental layer prior to a Maxent model run resulted in a higher-performing model (Radosavljevic and Anderson 2014). Therefore, it is possible that the AUC for each Maxent model prediction increased following the depth mask. The jackknife analysis of variable contributions finds that SST is the principal environmental predictor (36.9%), while pH contributes the least to the model prediction for present coral suitability (2.4%) (see Appendix, Table 10).

In both near-future and future scenarios under RCP 4.5, both difference maps indicate a general loss in suitable habitat across the Caribbean region (see Appendix, Figure 7-8). However, there is a gain in suitable habitat along the Florida Reef Tract, Mesoamerican Barrier Reef System (MBRS) along the Yucatan Peninsula, and in Cuba. The AUC for the two Maxent predictions under RCP 4.5 both increase to 0.720 and 0.714, respectively (see Appendix, Table 9).

While under RCP 8.5, each difference map suggests an overall loss in suitable habitat for all four reef-building coral families compared to present conditions (see Appendix, Figure 9-10). Although, areas along the southern tip of Florida and the Yucatan Peninsula experience an increase in suitable habitat. The AUC for near-future and future period Maxent predictions under RCP 8.5 increases to 0.722 and 0.718, respectively (see Appendix, Table 9). For all future time periods under both climate scenarios, sea surface temperature and salinity are the principal environmental predictors, while pH and PAR consistently influence the model the least (see Appendix, Table 11-14).

How each environmental layer influences the model greatly depends on the numeric change in the variable, in addition to the patterns and locations where these changes occur. For example, SST is expected to increase from a maximum of 29.8°C to 30.8°C under RCP 4.5 and 32.5°C under RCP 8.5 by 2080 (see Appendix, Table 8). Additionally, there is a heterogeneous change in SST throughout the region under each climate scenario (see Appendix, Figure 11). In contrast, pH is expected to undergo an overall change from a minimum of 8.2 to a minimum of 8.1 (RCP 4.5) and 7.9 (RCP 8.5) by 2080 (see Appendix, Table 8). However, under each scenario, there is a uniform shift in pH levels (see Appendix, Figure 12).

In regions where we expect a stabilization or increase in habitat suitability, environmental conditions include greater current velocities, higher pH, stable sea surface temperatures and photosynthetically active radiation levels, as well as less saline waters. All of these conditions correspond to the environmental factors coral reefs are restricted by today. As each of these environmental variables are influenced by climate change and shift beyond coral tolerance thresholds, we see a decrease in suitable habitat.

Discussion

Model Outcomes

The bioclimatic modelling results indicate that climate change will impact the distribution of suitable coral reef habitat in the future. However, we note an overall shift rather than absolute loss of suitable coral habitat across the entire Caribbean, specifically in regions with existing coral reefs such as the Florida Reef Tract on the southern end of Florida and the MBRS on the coast of Mexico's Yucatan Peninsula down to Belize and Guatemala (see Appendix, Figure 7-10). Results indicate distinct regions of these large-spanning Caribbean reefs as suitable for future reef habitat, although suitability is limited to pre-existing reef areas and does not indicate habitat expansion in any future scenarios. Although this result differs with recent studies on coral reef suitability in the face of climate change (Couce et al. 2013; Freeman et al. 2013), our results align with the most recent study of coral distribution in the Caribbean (Freeman et al. 2015). This study found that high salinity, seasonal temperature range, and thermal stress drive the persistence of high suitability in the Caribbean.

In the Florida Reef Tract, we see a concentration of increased or stable suitability on the southernmost tip of Florida along Miami and the Florida Keys for both the near-future and future time steps in the RCP 4.5 and RCP 8.5 scenarios. Similarly, for the MBRS, we see a concentration of increased or stable suitability focused on the southeastern portion of the system along the coast of Belize and Guatemala. An opposite result is seen around island nations such as Cuba in all scenarios.

This shift in suitability may be explained by the changes in SST, salinity, or ocean current speeds (see Appendix, Figure 11, 13-14). Currents along the coast of the Florida Reef Tract and MBRS maintain an overall stable speed for coral suitability across time steps and climate scenarios; however along island nations such as Cuba, speeds are drastically reduced in those same time steps and scenarios. We'd like to note that sea level rise is a commonly cited threat to island nations, but is not a variable considered in our analysis during the Maxent run as past studies omit this variable as well (Burke et al. 2011; Freeman et al. 2015).

For all five scenarios (present, near future and future for both RCP 4.5 and RCP 8.5), the variables with the highest contribution to coral suitability are SST, salinity, and current speeds (see Appendix, Table 10-14). Salinity contributed close to 50% in 3 future climate conditions due to the high shift in the regime and limiting effects of salinity on coral growth and survival (see Appendix, Table 11-13) (Ferrier-Pages et al. 1999). The ability of current speeds to dissipate the adverse effects of increased or decreased SST and salinity concentrations within the water column may explain why these three variables are important environmental predictors in our model (McClanahan et al. 2005). Additionally, changes in SST, salinity, and current speeds are heterogeneous throughout the Caribbean region, which also may account for greater percent contributions compared to the other environmental variable inputs.

Alternatively, PAR and pH had low contributions to suitability. These specific results may be due to the minimal shift in the change of pH in the two climate scenarios (see Appendix, Table 8) and the restricted change of PAR in areas where there are few coral presence points (i.e., the northern end of Yucatan, Haiti/Dominican Republic, and the north end of Brazil) (see Appendix, Figure 15). This is likely a result of downscaling and interpolating the global model environmental variables to fill missing data gaps across the smaller Caribbean Islands, causing the absence of coastal marine ecosystem processes, such as waves and tides.

Additionally, our analysis applied the mean rather than the maximum or minimum of each environmental variable to Maxent. The Freeman et al. (2013) study attributes a large proportion of the loss in suitable habitat to the large shift in minimum aragonite saturation levels. However, our study, which used average pH as a proxy for aragonite saturation, found a difference of 0.3 in pH but a low contribution in each prediction. This difference in variable change may contribute to this difference in results.

Limitations & Opportunities for Future Analysis

For this analysis to inform the feasibility of restoration and management in the Caribbean, one must consider the following caveats previously noted:

1. Coral species selected for this analysis were chosen for their reef-building capacity. The rarity and/or endangered status of the species were not considered.

2. Maxent only considers a binary presence/absence of corals; other conditions of corals often reported such as health or percent cover were not considered.

3. Other conditions such as nutrient runoff, bleaching events, sea level rise, or disease beyond the 5 environmental variables were not considered in our analysis. However, these conditions have major influences on the prolonged ability of corals to exist in a specific region.

Recommendations

While considering the underlying biological and environmental conditions corals depend on, interested parties can prioritize specific areas in the Caribbean for further study and assessment of coral health, threats, and local support. Additionally, a near-future and future analysis across different climate scenarios allows investors to assess the realistic risk of directed restoration efforts.

In summary, this analysis suggests that the response of shallow, tropical, reef-building corals to climate change in two commonly used scenarios is not homogenous across the Caribbean when considering a suite of indicative variables known to influence coral growth and health. As such, the associated risk reduction potential and co-benefits to coastal communities also have variable response to climate change. In all time steps and climate scenarios, the southern tip of Florida along the Keys and the region adjacent to the Yucatan Peninsula, Belize, and Guatemala have the highest suitability in the face of climate change. Additional site-specific studies must be conducted to assess the feasibility of coral restoration in these regions.

Summary & Conclusion

As climate change and shifting habitat suitability threaten coral reefs, it is important to protect these ecosystems by encouraging investments in restoration. Natural infrastructure is unique in that it promotes coastal resilience, helping communities reduce storm damage while preserving habitat. Coral reefs are much more than an important natural defense against storms; they also provide valuable co-benefits, a portion of which is directly captured by coastal hotels in the form of increased revenue.

Our valuation of co-benefits can be used together with risk reduction data from TNC to enable hotels to make informed investment choices in preparation for a changing climate and more frequent and severe storms. The cost model we developed will help hotels project financial return on investments in coral reefs, which is a vital step in incentivizing restoration projects. Finally, our suitability maps will allow TNC to make informed decisions to ensure restoration projects implemented today will be successful in the future.

Our three project objectives are designed to be replicable and applied to any location. We found that the hotel industry can be incentivized to invest in restoration given the large monetary benefit hotels receive from an additional meter of beach and an additional km^2 of coral, and given the low restoration cost we determined from expert interviews. We also found that there will be an increase in habitat suitability in all time steps and climate scenarios in certain regions of the Caribbean, particularly the southern tip of Florida along the Keys, the Yucatan Peninsula, Belize, and Guatemala. Our co-benefit valuation approach, cost model, and suitability analysis will allow TNC to identify areas with promising investment opportunities for hotels that may result in coral reef restoration via an insurance mechanism. Additional analysis is needed to compare investments in natural infrastructure on an equal footing with other infrastructure investment options for hotels, i.e., hybrid and grey infrastructure.

References

Albert S, Leon JX, Grinham AR, Church JA, Gibbes BR, Woodroffe CD (2016) Interactions between sea level rise and wave exposure on reef island dynamics in the Solomon Islands. *Environmental Research Letters*, 11:054011. doi:10.1088/1748-9326/11/5/054011.

Allemand D, Tambutté E, Zoccola D, Tambutté S (2011) Coral calcification, cells to reefs. In: Dubinsky Z, Stambler N (eds) Coral reefs: An ecosystem in transition. Springer, Netherlands: 119–150.

Ainsworth TD, Heron SF, Ortiz JC, Mumby PJ, Grech A, Ogawa D, Eakin CK, Leggat W (2016) Climate change disables coral bleaching protection on the Great Barrier Reef. Science, 352(6283): 338-342.

Aronson R, Bruckner A, Moore J, Precht B, Weil E (2008) Acropora cervicornis. The IUCN Red List of Threatened Species. 2008: e.T133381A3716457. http://dx.doi.org/10.2305/IUCN.UK.2008.RLTS.T133381A3716457.en.

Aronson R, Bruckner A, Moore J, Precht B, Weil E (2008) Acropora palmata. The IUCN Red List of Threatened Species 2008: e.T133006A3536699. http://dx.doi.org/10.2305/IUCN.UK.2008.RLTS.T133006A3536699.en

Aronson, R., Bruckner A., Moore J., Precht B., Weil E (2008) Montastraea annularis. The IUCN Red List of Threatened Species 2008: e.T133134A3592972. http://dx.doi.org/10.2305/IUCN.UK.2008.RLTS.T133134A3592972.en.

Assis J, Tyberghein L, Bosh S, Verbruggen H, Serrão EA, De Clerck O (2017) Bio-ORACLE v2.0: Extending marine data layers for bioclimatic modelling. *Global Ecology and Biogeography*.

Barnes RD (1987) Invertebrate Zoology; Fifth Edition. Fort Worth, TX: *Harcourt Brace Jovanovich College Publishers*: 92-96, 127-134, 149-162.

Basher Z, Bowden DA, Costello MJ (2018) GMED: Global Marine Environment Datasets for environment visualisation and species distribution modelling. *Earth Systems Science Data*, https://doi.org/10.5194/essd-2018-64.

Bayraktarov, Elisa, et al. (2016) "The Cost and Feasibility of Marine Coastal Restoration. *Ecological Applications*, vol. 26, no. 4, pp. 1055–74. *Crossref*, doi:10.1890/15-1077.

Beck MW, Losada IJ, Menendez P, Reguero BG, Diaz-Simal P, Fernandez F (2018) The global flood protection savings provided by coral reefs. *Nature Communications* 9: 2186. DOI: 10.1038/s41467-018-04568-z.

Bruno JF, Valdivia A (2016) Coral reef degradation is not correlated with local human population density. Scientific Reports: 6.

Bulleri, F and Chapman, MG. (2010) "The Introduction of Coastal Infrastructure as a Driver of Change in Marine Environments." *Journal of Applied Ecology*, 47(1): 26–35. doi:10.1111/j.1365-2664.2009.01751.x.

Burke L, Reytar K, Spalding M, Perry A. (2011) Reefs at risk revisited. World Resources Institute: Washington, DC. ISBN 978-1-56973-762-0. 115.

Burke, L, Maidens J. (2004) Reefs at Risk in the Caribbean. World Resources Institute: Washington, DC. ISBN 1-56973-567-0. 8.

Clarke L, Edmonds J, Jacoby H, Pitcher H, Reilly J, Richels R (2007) CCSP synthesis and assessment product 2.1, Part A: scenarios of greenhouse gas emissions and atmospheric concentrations. U.S. Government Printing Office, Washington, DC.

Comeau S, Edmunds PJ, Lantz CA, Carpenter RC (2014) Water flow modulates the response of coral reef communities to ocean acidification. *Scientific reports*, 4(6681), doi:10.1038/srep06681.

Couce E, Ridgwell A, Hendy EJ (2012) Environmental controls on the global distribution of shallow-water coral reefs. *Journal of Biogeography*, 39: 1508–1523.

D'Angelo C, Wiedenmann J (2014) Impacts of nutrient enrichment on coral reefs: new perspectives and implications for coastal management and reef survival. *Current Opinion in Environmental Sustainability*, 7: 82-93, https://doi.org/10.1016/j.cosust.2013.11.029.

Deloitte Access Economics (2017) "At what price? The economic, social, and icon value of the Great Barrier Reef." Brisbane, Australia.

Durack PJ, Wijffels SE (2010) Fifty-Year Trends in Global Ocean Salinities and Their Relationship to Broad-Scale Warming. *Journal of Climate*, 23: 4342–4362, https://doi.org/10.1175/2010JCLI3377.1

Elith J, Graham CH, Anderson RP (2006) Novel methods improve prediction of species' distributions from occurrence data. *Ecography*, 29: 129–151.

Elith J, Leathwick JR, Hastie T (2008) A Working Guide to Boosted Regression Trees. *Journal of Animal Ecology*, 77(4): 802–13. *Wiley Online Library*, doi:10.1111/j.1365-2656.2008.01390.x.

Fabricius KE, Langdon C, Uthicke S, Humphrey C, Noonan S, De'ath G, Okazaki R, Muehllehner N, Glas MS, Lough JM (2011) Losers and winners in coral reefs acclimatized to elevated carbon dioxide concentrations. *Nature Climate Change* 1: 165-169.

Ferrario F, Beck MW, Storlazzi CD, Micheli F, Shepard CC, Airoldi L (2014) The Effectiveness of Coral Reefs for Coastal Hazard Risk Reduction and Adaptation. *Nature Communications*, 5. *Crossref*, doi:10.1038/ncomms4794.

Ferrier-Pages C, Gattuso J, Jaubert J (1999) Effect of small variations in salinity on the rates of photosynthesis and respiration of the zooxanthellate coral Stylophora pistillata. *Marine Ecology-Progress Series,* 181: 309-314.

Fisher B, Nakicenovic N, Alfsen K, Corfee Morlot J, De la Chesnaye F, Hourcade J.-C, Jiang K, Kainuma M, La Rovere E, Matysek A, Rana A, Riahi K, Richels R, Rose S, van Vuuren DP, Warren R, (2007) Chapter 3: Issues related to mitigation in the long-term context. In: Climate change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA

Franklin J (2009) Mapping species distributions: spatial inference and prediction. Cambridge University Press, Cambridge, UK.

Freeman LA, Kleypas JA, Miller AJ (2013) Coral reef habitat response to climate change scenarios. *PLoS ONE,* 8(12): e82404. doi:10.1371/journal.pone.0082404.

Freeman AL (2015) Robust Performance of Marginal Pacific Coral Reef Habitats in Future Climate Scenarios. *PLoS One*. s://doi.org/10.1371/journal.pone.0128875

Gomes V, Ijff S, Raes N, Amaral Ieda, Salomão RP, Coêlho L, Matos F, Castilho C, Filho DALF, López DC, Guevara J, Magnusson W, Phillips O, Wittmann F, Carim MJV, Martins MP, Irume M, Sabatier D, Molino JF, ter Steege H (2018) Species Distribution Modelling: Contrasting presence-only models with plot abundance data. *Scientific Reports,* 8(1003). 10.1038/s41598-017-18927-1.

Goreau TJ, Hilbertz W (2005) Marine Ecosystem Restoration: Costs and Benefits for Coral Reefs. *World Resource Review,* 17: 375-409.

Gourlay MR (1997) 13th Australasian Coastal and Ocean Engineering Conference & the 6th Australasian Port and Harbour Conference. Centre for Advanced Engineering, University of Canterbury, *Pacific Coasts and Ports '97: Proceedings of the 13th Australasian Coastal and Ocean Engineering Conference and the 6th Australasian Port and Harbour Conference,* 927–932.

Graham NAJ, McClanahan TR, MacNeil MA, Wilson SK, Polunin NVC, Jennings S, Chabanet P, Clark S, Spalding MD, Letourneur Y, Bigot L, Galzin R, Öhman MC, Garpe KC, Edwards AJ, Sheppard CRC (2008) Climate warming, marine protected areas and the ocean-scale integrity of coral reef ecosystems. *PLoS ONE,* 3:e3039.

Guannel G, Arkema K, Ruggiero P, Verutes G (2016) The Power of Three: Coral Reefs, Seagrasses and Mangroves Protect Coastal Regions and Increase Their Resilience. PLoS ONE 11(7): e0158094. https://doi.org/10.1371/journal.pone.0158094.

Hinkel J, Klein RJT (2013) Coastal flood damage and adaptation costs under 21st century sea-level rise. *PNAS, Proceedings of the National Academy of Sciences*, 111(9):3292–3297. doi: 10.1073/ pnas.1222469111

Hinkel J, Lincke D, Vafeidis AT, Perrette M, Nicholls RJ, Tol RSJ, Marzeion B, Fettweis X, Ionescu C, Levermann A (2014) Coastal Flood Damage and Adaptation Costs under 21st Century Sea-Level Rise. *Proceedings of the National Academy of Sciences*, 111(9): 3292–97. *Crossref*, doi:10.1073/pnas.1222469111.

Hoegh-Guldberg O (1999) Climate change, coral bleaching and the future of the world's coral reefs. *Marine Freshwater Resources*, 50: 839-66.

IPCC (2008) Towards new scenarios for analysis of emissions, climate change, impacts, and response strategies. IPCC Expert Meeting Report on New Scenarios, Noordwijkerhout, Intergovernmental Panel on Climate Change

IPCC (2014) Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. IPCC, Geneva, Switzerland, 151 pp.

Kay JE, Deser C, Phillips A, Mai A, Hannay C, Strand G, Arblaster J, Bates S, Danabasoglu G, Edwards J, Holland M, Kushner P, Lamarque JF, Lawrence D, Lindsay K, Middleton A, Munoz E, Neale R, Oleson K, Polvani L, Vertenstein M (2015), The Community Earth System Model (CESM) Large Ensemble Project: A Community Resource for Studying Climate Change in the Presence of Internal Climate Variability, Bulletin of the American Meteorological Society, doi: 10.1175/BAMS-D-13-00255.1, 96: 1333-1349

Emillson T, Sang AO (2017) "Impacts of climate change on urban areas and nature-based solutions for adaptation," in *Nature-based Solutions to Climate Change Adaptation in Urban Areas, Theory and Practice of Urban Sustainability Transitions*, eds N. Kabisch, H. Korn, J. Stadler, A. Bonn (Cham: Springer), 15-27. doi:10.1007/978-3-319-56091-5_2.

Kleypas JA, Kimberly KYK (2009) Coral Reefs and Ocean Acidification. *Oceanography*, 22(4): 108-117. Retrieved from http://www.jstor.org/stable/24861028.

Latinopoulos D (2018) Using a Spatial Hedonic Analysis to Evaluate the Effect of Sea View on Hotel Prices. *Tourism Management*, 65: 87–99. ScienceDirect, doi:10.1016/j.tourman.2017.09.019.

Lunden JJ, Georgian SE, Cordes EE (2013) Aragonite saturation states at cold-water coral reefs structured by *Lophelia pertusa* in the northern Gulf of Mexico. *Limnology and Oceanography* 58(1): 354-362.

Marsh ND, Svensmark H (2000) Low cloud properties influenced by cosmic rays. *Physical Review Letters,* 85(23): 5004–5007. doi:https://doi.org/10.1103/PhysRevLett.85.5004

McClanahan, TR, Ateweberhan M, Muhando CA, Maina J, Mohammed MS (2007) Effects of Climate and Seawater Temperature Variation on Coral Bleaching and Mortality. *Ecological Monographs*, 77: 503-525. doi:10.1890/06-1182.1.

McGranahan G, Balk D, Anderson B (2007) The rising tide: assessing the risks of climate change and human settlements in low elevation coastal zones. *Environment and Urbanziation*, 19: 17-37.

McLeod KB (2016) "Climate Risk & Resilience: Recognizing the protective values of nature" [Fact sheet].

McManus JW, Polsenberg JF (2004) Coral–algal phase shifts on coral reefs: ecological and environmental aspects. *Progress in Oceanography*, 60(2): 263–279.

McQuarrie ME, Pilkey OH (1998) Evaluation of Alternative or Non-Traditional Shoreline Stabilization Devices. *Journal of Coastal Research*, 269–72.

Merow C, Smith MJ, Silander JA (2013) A practical guide to MaxEnt for modeling species' distributions: what it does, and why inputs and settings matter. *Ecography*, 36: 1058–1069.

Moberg F, Folke C (1999) Ecological goods and services of coral reef ecosystems. *Ecological Economics*, *29*(2), 215–233. https://doi.org/10.1016/S0921-8009(99)00009-9.

Monks, Ellory. (2017) "Cities Can't Prejudge Winner in Green v. Grey Infrastructure Battle." *Meeting of the Minds* http://meetingoftheminds.org/cities-cant-prejudge-winner-green-v-grey-infrastructure-battle-21621

NOAA National Marine Fisheries Service. "General fact sheet Atlantic *Acropora* corals." https://sero.nmfs.noaa.gov/protected_resources/outreach_and_education/documents/acropora_factsheet.pdf

Narayan S, Beck MW, Reguero BG, Losada IJ, Wesenbeeck BV, Pontee N, Sanchirico JN, Ingram JC, Lange GM, Burks-Copes K (2016) The effectiveness, costs and coastal protection benefits of natural and nature-based defenses. *PLoS ONE,* 11(5): e0154735. doi:10.1371/ journal.pone.0154735.

Neumann B, Vafeidis AT, Zimmermann J, Nicholls RJ (2015) Future Coastal Population Growth and Exposure to Sea-Level Rise and Coastal Flooding—A Global Assessment. *PLoS ONE* 10(3): e0118571.

Nicholls RJ, Cazenave A (2010) Sea-level rise and its impact on coastal zones. *Science,* 328(5985): 1517-1520. doi: 10.1126/science.1185782.

Nordstrom, Karl F. (2014) "Living with Shore Protection Structures: A Review." *Estuarine, Coastal and Shelf Science*, vol. 150, pp. 11–23. *Crossref*, doi:10.1016/j.ecss.2013.11.003.

Pandolfi JM, Connolly SR, Marshall DJ, Cohen AL (2011) Projection Coral Reef Futures Under Global Warming and Ocean Acidification. *Science,* 333(6041): 418-422.

Perlman H (2017) "The 100-Year Flood-It's All About Chance (USGS)." U.S. Geological Survey, water.usgs.gov/edu/100yearflood-basic.html.

Phillips SJ, Anderson RP, Schapire RE (2006) Maximum entropy modeling of species geographic distributions. *Ecological Modelling*, 190: 231–259.

Phillips SJ, Dudík M (2008) Modeling of species distributions with Maxent: new extensions and a comprehensive evaluation. *Ecography*, 31: 161-175. doi:10.1111/j.0906-7590.2008.5203.x

Phillips SJ, Dudík M, Elith J, Graham CH, Lehmann A, Leathwick J, Ferrier S (2009) Sample selection bias and presence-only distribution models: implications for background and pseudo-absence data. *Ecological Applications*, 19: 181–197.

"Places to Go." *Visit Florida*, https://www.visitflorida.com/en-us/places-to-go.html#SE. Accessed 21 Mar. 2019.

Putnam HM, Barott KL, Ainsworth TD, Gates RD. (2017) The Vulnerability and Resilience of Reef-Building Corals. *Current Biology*, 27(11). https://doi.org/10.1016/j.cub.2017.04.047.

Radosavljevic A, Anderson RP (2014) Making better Maxent models of species distributions: complexity, overfitting and evaluation. *Journal of Biogeography*, 41: 629-643. doi:10.1111/jbi.12227.

Rahmstorf S (2017) Rising hazard of storm-surge flooding. Proceeding National Academy Science. 114 11806–8

Reed, M. (2003, June 7).Why businesses should pay to protect ecosystems. Retrieved from https://www.theguardian.com/sustainable-business/businesses-pay-protect-ecosystems.

Reguero BG, Losada IJ, Diaz-Simal P, Mendez FJ, Beck MW (2015) Effects of climate change on exposure to coastal flooding in Latin America and the Caribbean. *PLoS ONE*, 10(7): e0133409. doi:10.1371/journal.pone.0133409.

Reguero BG, Beck MW, Bresch DN, Calil J, Meliane I (2018) Comparing the cost effectiveness of nature-based and coastal adaptation: A case study from the Gulf Coast of the United States. PLoS ONE, 13(4): e0192132. https://doi.org/10.1371/journal.pone.0192132

Reguero BG, Beck MW, Agostini VN, Kramer P, Hancock B (2018) Coral reefs for coastal protection: A new methodological approach and engineering case study in Grenada. *Journal of Environmental Management*, 210: 146-161.

Reguero BG, Losada IJ, Mendez FJ (2019) A recent increase in global wave power as a consequence of oceanic warming. *Nature Communications* 10: 205. doi: 10.1038/s41467-018-08066-0.

Riahi K, Rao S. Krey V, Cho C, Chirkov V, Fischer G, Kindermann G, Nakicenovic N, Rafaj P (2011) RCP 8.5 - A scenario of comparatively high greenhouse gas emissions. *Climatic Change,* 109: 33-57.

Rigall-I-Torrent R, Fluvia M, Ballester R, Salo A, Ariza E, Espinet JM (2011) The Effects of Beach Characteristics and Location with Respect to Hotel Prices. *Tourism Management*, 32(5): 1150–1158, doi:https://doi.org/10.1016/j.tourman.2010.10.005.

Selig ER, Casey KS, Bruno JF (2012) Temperature-driven coral decline: the role of marine protected areas. *Global Change Biology*, 18: 1561-1570.

Sheppard C, Dixon DJ, Gourlay M, Sheppard A, Payet R (2005) Coral mortality increases wave energy reaching shores protected by reef flats: Examples from the Seychelles. *Estuarine, Coastal and Shelf Science*, 64: 223-234.

Silva M, MacDonald IR (2017) Habitat suitability modeling for mesophotic coral in the northeastern Gulf of Mexico. *Marine Ecology Progress Series*, 583: 121-136.

Slater H, Michael E (2012) Predicting the current and future potential distributions of lymphatic filariasis in Africa using maximum entropy ecological niche modelling. *PLoS ONE*, 7(2): e32202.

Soler IP, Gemar G, Correia MB, Serra F (2019) Algarve Hotel Price Determinants: A Hedonic Pricing Model. *Tourism Management*, 70: 311–21. *ScienceDirect*, doi:10.1016/j.tourman.2018.08.028.

Soler, Ismael P., and German Gemar. “Hedonic Price Models with Geographically Weighted Regression: An Application to Hospitality.” *Journal of Destination Marketing & Management*, vol. 9, Sept. 2018, pp. 126–37. *ScienceDirect*, doi:10.1016/j.jdmm.2017.12.001.

Spalding MD, Ruffo S, Lacambra C, Meliane I, ale LZ, Sheprd CC, Beck MW (2014) The role of ecosystems in coastal protection: Adapting to climate change and coastal hazards. *Ocean & Coastal Management*, 90: 50-57.

Spalding M, Burke L, Wood SA, Ashpole J, Hutchison J, Ermgassen Pz (2017) Mapping the global value and distribution of coral reef tourism. *Marine Policy*, 82: 104-113. doi: 10.1016/j.marpol.2017.05.014.

Spurgeon JPG, Lindahl U (2000) Economics of coral reef restoration. In Cesar, Herman S. J., and Cordio, Department for Biology and Environmental Sciences, editors. *Collected Essays on the Economics of Coral Reefs*, 124-37.

Steinberg C (2007) Impacts of climate change on the physical oceanography of the Great Barrier Reef. In: *Climate Change and the Great Barrier Reef* (eds J.E. Johnson andP.A. Marshall). Great Barrier Reef Marine Park Authority and Australian Greenhouse Office, Townsville, Qld, pp. 51– 74.

Sutton-Grier AE, Wowk K, Bamford H (2015) Future of Our Coasts: The Potential for Natural and Hybrid Infrastructure to Enhance the Resilience of Our Coastal Communities, Economies and Ecosystems. *Environmental Science & Policy*, 51: 137–48. *Crossref*, doi:10.1016/j.envsci.2015.04.006.

Rosen S (1974) Hedonic Prices and Implicit Markets: Product Differentiation in Pure Competition. *Journal of Political Economy*, 82(1): 34. *EBSCOhost*, doi:10.1086/260169.

The Nature Conservancy. The Coastal Zone Management Trust: Quintana Roo, Mexico. https://www.nature.org/content/dam/tnc/nature/en/documents/TNC_Mexico_CoastalManagementTrust_Factsheet.pdf.

Thomson AM, Calvin KV, Smith SJ, Kyle GP, Volke A, Patel P, Delgado-Arias S, Bond-Lamberty B, Wise MA, Clarke LE (2011) RCP4.5: a pathway for stabilization of radiative forcing by 2100. *Climatic Change*. doi: 10.1007/s10584-011-0151-4.

Tyberghein L, Verbruggen H, Pauly K, Troupin C, Mineur F, De Clerck O (2012) Bio-ORACLE: A global environmental dataset for marine species distribution modelling. *Global Ecology and Biogeography*, 21: 272-281.

van Zanten BT, van Beukering PJH, Wagtendonk AJ (2014) Coastal Protection by Coral Reefs: A Framework for Spatial Assessment and Economic Valuation. *Ocean & Coastal Management*, 96: 94–103. *ScienceDirect*, doi:10.1016/j.ocecoaman.2014.05.001.

Weber WL (2014) Production, Growth, and the Environment: an Economic Approach. CRC Press/Taylor and Francis Group.

Wells, Sue and Ravilious, Corinna. *In the Front Line: Shoreline Protection and Other Ecosystem Services from Mangroves and Coral Reefs*. UNEP World Conservation Monitoring Centre, 2006. Google books. Web. 20 Mar. 2019.

Wielgus JE, Cooper E, Torres R, Burke L (2010) Coastal Capital: Dominican Republic. Case studies on the economic value of coastal ecosystems in the Dominican Republic. Working Paper. Washington, DC: World Resources Institute. Available online at http://www.wri.org/ coastal-capital.

Winkler NS, Pandolfi JM, Sampayo EM (2014) *Symiodinium* identity alters the temperature-dependent settlement behavior of *Acropora millepore* coral larvae before the onset of symbiosis. *Proceedings Royal Society B,* 282: 20142260. doi: 10.1098/rspb.2014.2260.

Winters R and Festa J (2018) “A First-of-Its-Kind, Multi-Sector Partnership Will Protect People, Their Livelihoods, and Nature along Mexico's #Reefs #OceanSummit.” *The Nature Conservancy*, www.nature.org/en-us/explore/newsroom/the-nature-conservancy-and-the-government-of-quintana-roo-announce-innovativ/.

Appendix

Table 8. Summary statistics (mean, minimum, maximum, and the difference from observed present values) for each environmental layer in present conditions, and near-future and future time periods under climate scenarios RCP 4.5 and RCP 8.5.

Variable	Mean	Minimum	Maximum
Present (2008 - 2018) - Variable Summary Statistics			
Current Speeds (cm/s)	10.28	0.02	130.87
Salinity (ppt)	35.6	18.64	37.16
SST (°C)	27.55	23.23	29.78
PAR (W/m²)	43.12	32	50
pH	8.19	7.92	8.28
RCP 4.5 (2040 - 2050) - Variable Summary Statistics			
Current Speeds (cm/s)	9.4	-1.24	98.32
Salinity (ppt)	35.81	23.08	37.3
SST (°C)	28.12	24.29	30.23
PAR (W/m²)	43.72	32.88	49.26
pH	8.13	7.87	8.22
RCP 4.5 (2070 - 2080) - Variable Summary Statistics			
Current Speeds (cm/s)	9.36	-1.23	97.06

Salinity (ppt)	35.93	23.12	37.48
SST (°C)	28.62	25.1	30.84
PAR (W/m²)	44.11	33.19	49.4
pH	8.09	7.83	8.12
RCP 8.5 (2040 - 2050) - Variable Summary Statistics			
Current Speeds (cm/s)	9.28	-1.21	99.02
Salinity (ppt)	35.87	23.11	37.38
SST (°C)	28.38	24.54	30.52
PAR (W/m²)	43.87	32.5	49.22
pH	8.1	7.84	8.19
RCP 8.5 (2070 - 2080) - Variable Summary Statistics			
Current Speeds (cm/s)	9.48	-47.35	118.64
Salinity (ppt)	36.36	22.45	38.08
SST (°C)	30.2	26.77	32.54
PAR (W/m²)	44.2	30.56	49.14
pH	7.91	7.65	7.99

Table 9. Performance of *Acroporidae, Montastraeidae, Merulinidae*, and *Poritidae* coral family distribution models measured by cross-validated area-under-curve (AUC) for Maxent model predictions of present conditions and near-future and future time periods under RCP 4.5 and 8.5.

Scenario	AUC
Present (2008 - 2018)	0.722
RCP 4.5 (2040 - 2050)	0.720
RCP 4.5 (2070 - 2080)	0.714
RCP 8.5 (2040 - 2050)	0.722
RCP 8.5 (2070 - 2080)	0.718

Table 10. Jackknife analysis of variable contributions to the present (2008 - 2018) Maxent prediction of suitable habitat for *Acroporidae, Montastraeidae, Merulinidae*, and *Poritidae* coral families.

Present (2008 - 2018) - Analysis of Variable Contributions:		
Variable	**Percent contribution**	**Permutation importance**
SST	36.9	46
Salinity	35.5	34.7
Current speeds	16.8	6.4
PAR	8.3	11.9
pH	2.4	1

Table 11. Jackknife analysis of variable contributions to the near future (2040 - 2050) RCP 4.5 Maxent *Acroporidae, Montastraeidae, Merulinidae*, and *Poritidae* coral families.

RCP 4.5 (2040 - 2050) Scenario - Analysis of Variable Contributions:		
Variable	**Percent contribution**	**Permutation importance**
Salinity	47.6	40.4
SST	31.6	44.4
Current speeds	18.8	14.4
pH	1	0.1
PAR	0.9	0.7

Table 12. Jackknife analysis of variable contributions to the future (2070 - 2080) RCP 4.5 Maxent *Acroporidae, Montastraeidae, Merulinidae*, and *Poritidae* coral families.

RCP 4.5 (2070 - 2080) Scenario - Analysis of Variable Contributions:		
Variable	**Percent contribution**	**Permutation importance**
Salinity	47.1	42.5
SST	32.6	43
Current speeds	18.8	13.2
pH	0.9	1.1
PAR	0.7	0.2

Table 13. Jackknife analysis of variable contributions to the near future (2040 - 2050) RCP 8.5 Maxent *Acroporidae, Montastraeidae, Merulinidae*, and *Poritidae* coral families.

RCP 8.5 (2040 - 2050) Scenario - Analysis of Variable Contributions:		
Variable	**Percent contribution**	**Permutation importance**
Salinity	47.4	40.9
SST	32.4	46.1
Current speeds	17.6	12.4
pH	2	0
PAR	0.4	0.5

Table 14. Jackknife analysis of variable contributions to the future (2070 - 2080) RCP 8.5 Maxent *Acroporidae, Montastraeidae, Merulinidae*, and *Poritidae* coral families.

RCP 8.5 (2070 - 2080) Scenario - Analysis of Variable Contributions:		
Variable	**Percent contribution**	**Permutation importance**
SST	37.4	38.5
Salinity	32.3	38.1
Current speeds	18.1	19.7
pH	9	2.9
PAR	3.3	0.8

Reef Restoration Cost Build-Up

Model Inputs

Reef project parameters	
Height (m)	1
Width (m)	1000
Depth (m)	1
Distance to nursery (miles)	20
Nursery care	
Nursery hard costs ($/m3)	$8
Veterinary checkups ($/m3)	$1
Project preparation	
Project engineering	$150,000
Permitting	$30,000
Boat fuel and expenses ($/mile)	$150
Reef materials ($/m^3)	
Structural hard costs ($/m^3)	$150
Coral outplantings hard costs ($/m^2)	$15
Labor	
Grading ($/m^2)	$5
Staff training	10%
Nursery care labor	75%
Structure installation labor	75%
Coral outplantings installation	75%
Project management (% total)	20%
Post implementation costs	
Monitoring costs ($/year)	$1,500
Maintenance costs ($/year)	$5,000
Total years	15

High-Level Budget

Preparation phase	
Project engineering	150,000
Permits	30,000
Nursery materials	9,000
Nursery care labor costs	6,750
Total	195,750
Implementation phase	
Grading	5,000
Structural hard costs	150,000
Structure installation labor costs	112,500
Boat fuel and expenses	3,000
Coral outplantings	15,000
Coral installation labor costs	11,250
Subtotal	296,750
Staff training	29,675
Project management	59,350
Total	385,775
Post-Implementation Phase	
Monitoring	22,500
Maintenance	75,000
Total	97,500
Grand Total	**679,025**
Cost per km	*679,025*

Figure 4. Cost of reef restoration mini-model. The smaller components of this model can be filled in by restoration groups to help determine the overall cost of restoration. The current numbers and percentages are rough estimates based on interviews, and serve as place holders.

Questions on each individual project

1. Project location and year
2. Project size
3. Was is the problem you are trying to address with the project (habitat restoration, species recovery, wave attenuation, etc.)?
4. What are the key steps/ components in the project?
5. How did you grow your coral in the nursery?
6. Does the project include an artificial component (e.g. cement substrate)?
7. What are the key components that determine the total cost?
8. What are the labor and material costs?
9. What are the maintenance costs?
10. Were volunteers involved?
11. Were permitting requirements a barrier?
12. Who is funding the work?

General questions

13. What makes up the large spatial variations of cost in different areas?
14. How has the field of restoration changed in the previous 10 years?

Figure 5. Cost Questionnaire for Restoration Professionals. A list of questions that were asked to each of the interviewees from reef restoration organizations.

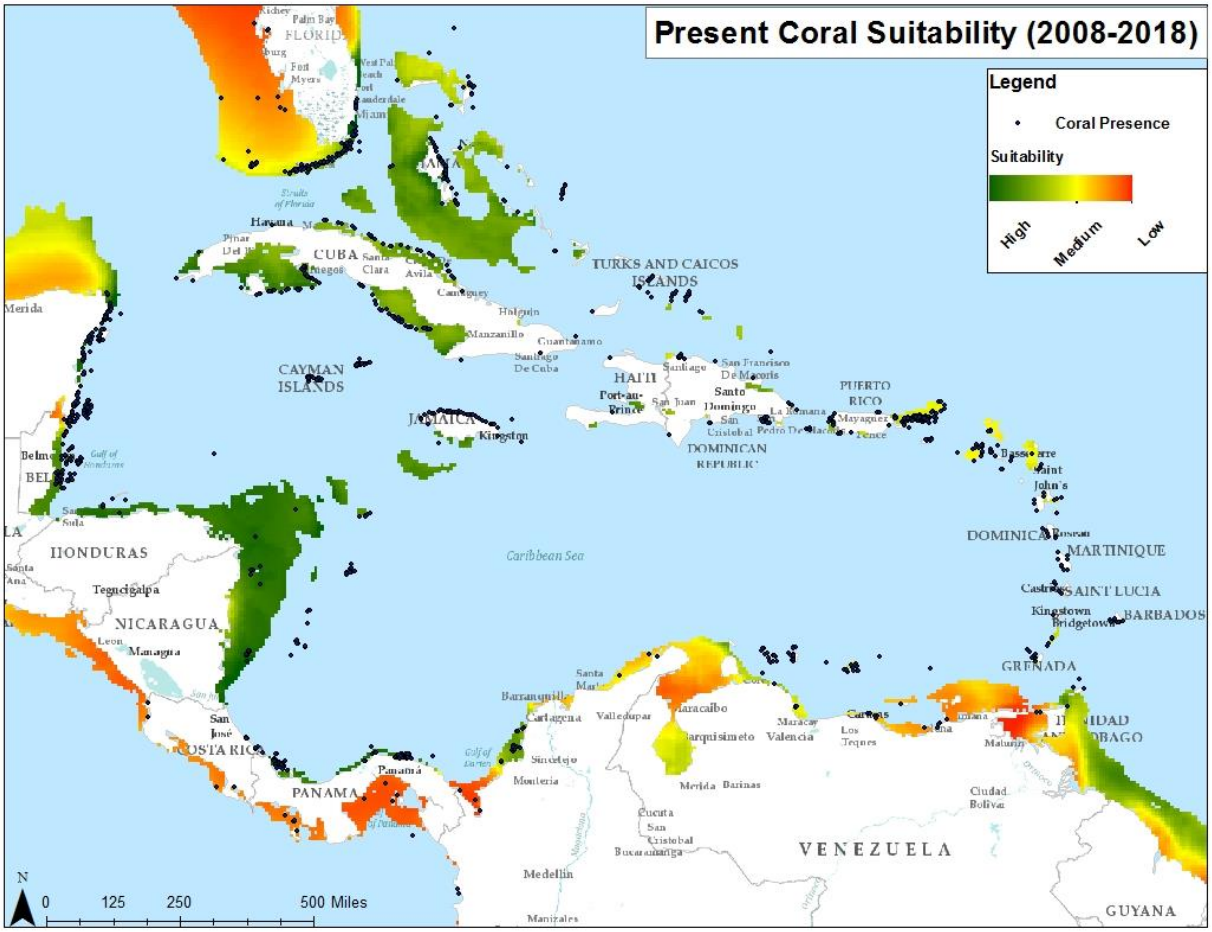

Figure 6. Present suitable habitat (2008 – 2018) for *Acroporidae, Montastraeidae, Merulinidae*, and *Poritidae* coral families. Color scale indicates the probability of suitable habitat for coral reefs. Green areas indicate probability of high suitability, yellow areas indicate probability of medium suitability, and red areas indicate probability of low suitability. This region of study is bounded by longitudes ranging from 103°W to 50°W, and latitudes ranging from 30°N to 4°S.

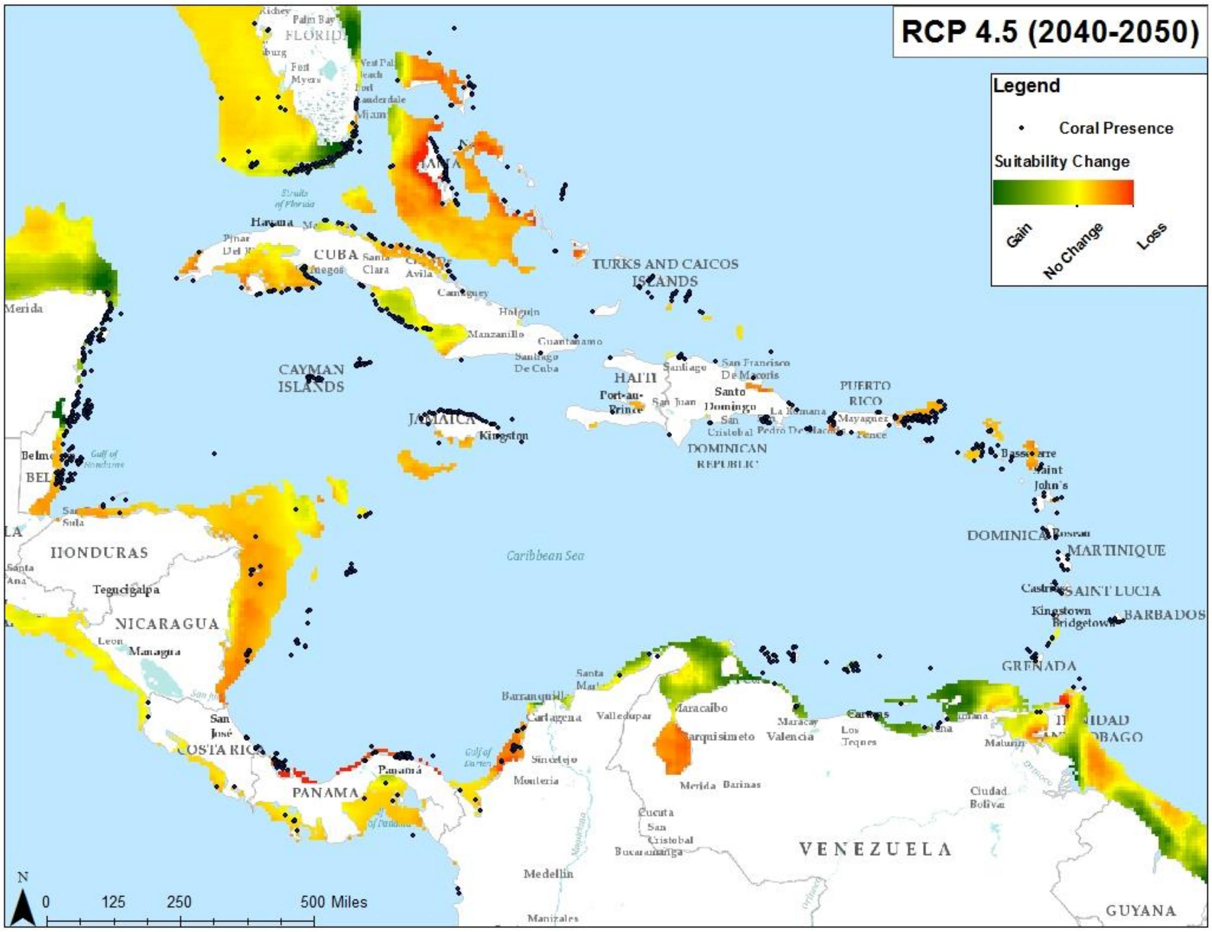

Figure 7. Difference in coral reef suitable habitat in the near future (2040-2050) under RCP 4.5 compared to present conditions for *Acroporidae, Montastraeidae, Merulinidae*, and *Poritidae* coral families. Color scale indicates a gain or loss in the probability of suitable habitat for coral reefs. Green areas indicate an increase in the probability of high suitability, yellow areas suggest little to no change in the probability of medium suitability, and red areas indicate a decrease in the probability of low suitability. This region of study is bounded by longitudes ranging from 103°W to 50°W, and latitudes ranging from 30°N to 4°S.

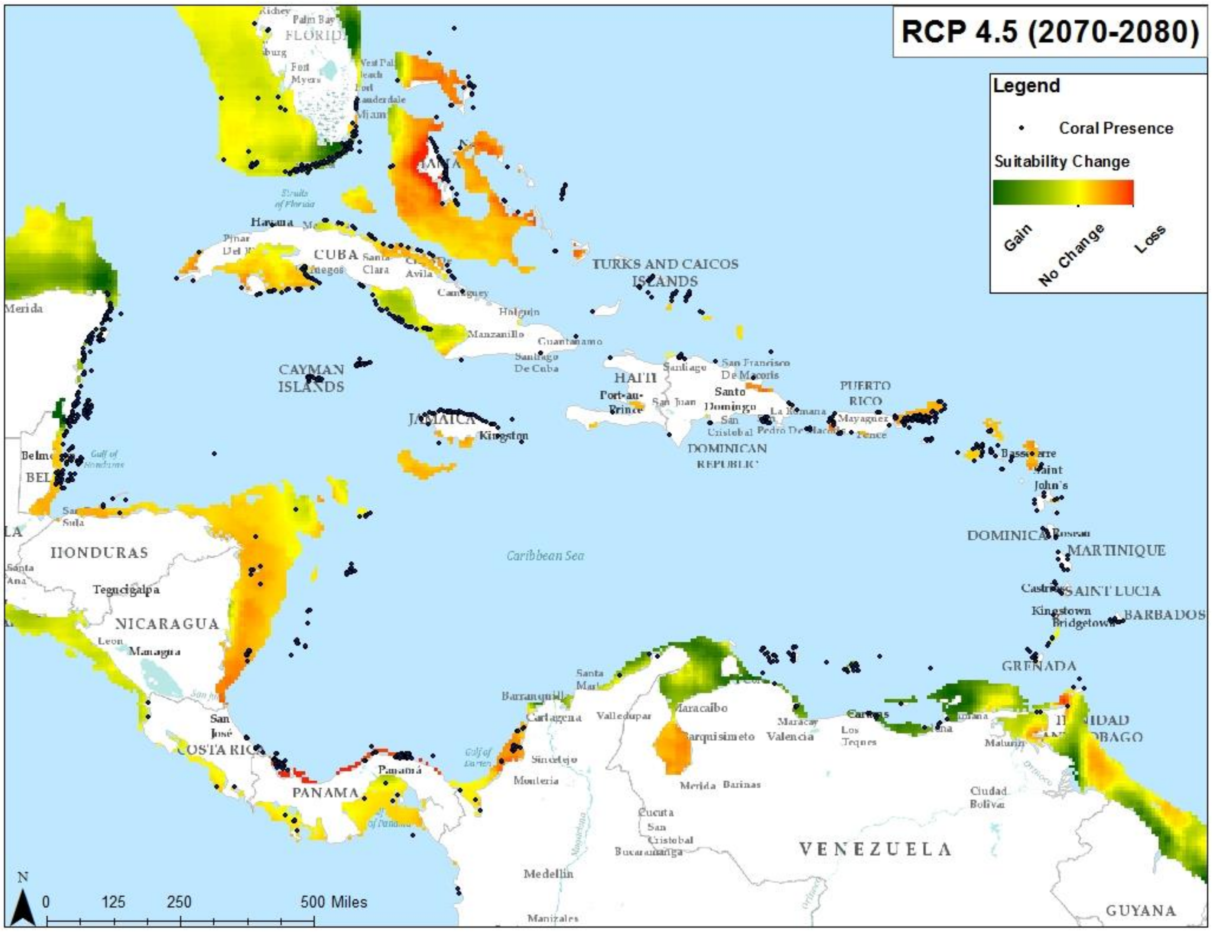

Figure 8. Difference in coral reef suitable habitat in the future (2070-2080) under RCP 4.5 compared to present conditions for *Acroporidae, Montastraeidae, Merulinidae*, and *Poritidae* coral families. Color scale indicates a gain or loss in the probability of suitable habitat for coral reefs. Green areas indicate an increase in the probability of high suitability, yellow areas suggest little to no change in the probability of medium suitability, and red areas indicate a decrease in the probability of low suitability. This region of study is bounded by longitudes ranging from 103°W to 50°W, and latitudes ranging from 30°N to 4°S.

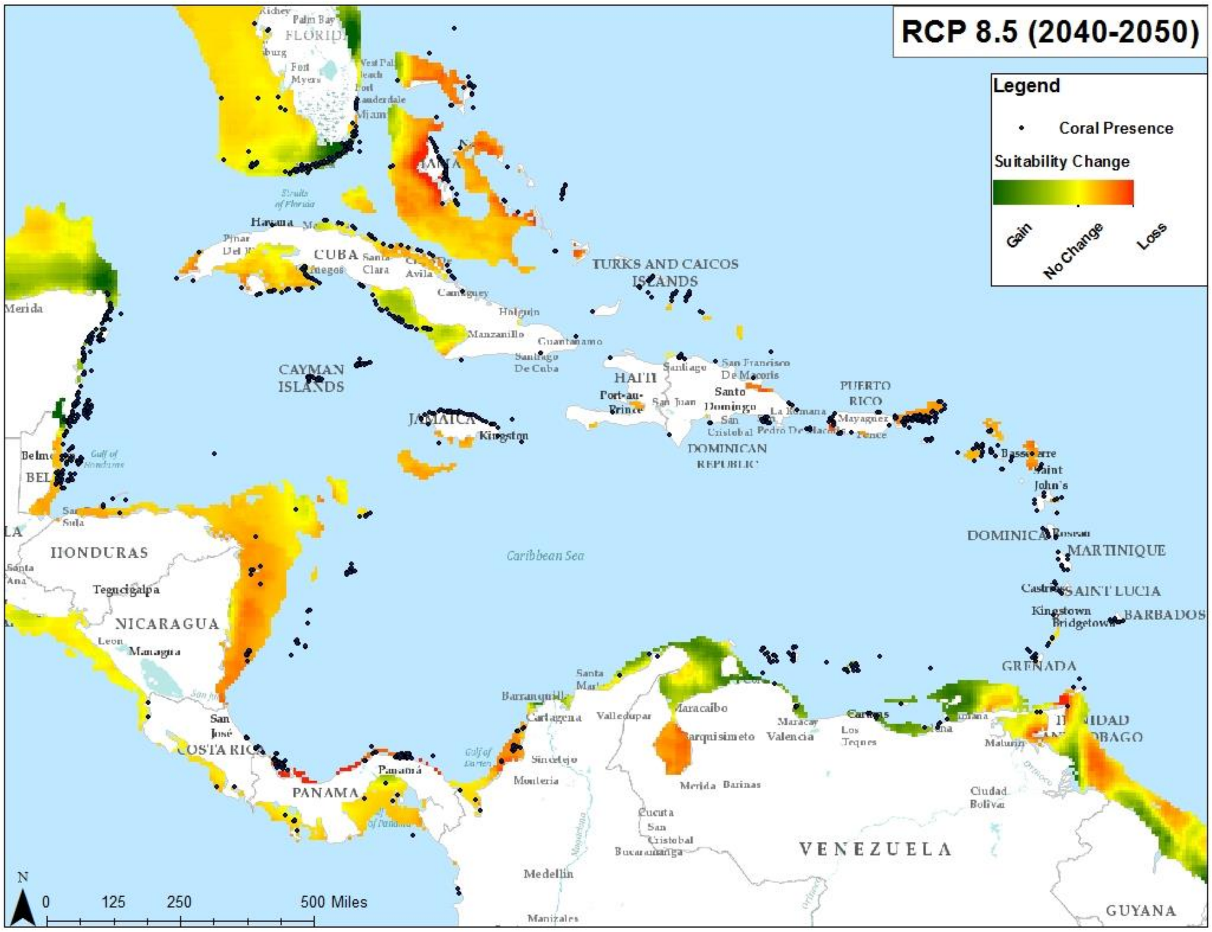

Figure 9. Difference in coral reef suitable habitat in the near future (2040-2050) under RCP 8.5 compared to present conditions for *Acroporidae, Montastraeidae, Merulinidae*, and *Poritidae* coral families. Color scale indicates a gain or loss in the probability of suitable habitat for coral reefs. Green areas indicate an increase in the probability of high suitability, yellow areas suggest little to no change in the probability of medium suitability, and red areas indicate a decrease in the probability of low suitability. This region of study is bounded by longitudes ranging from 103°W to 50°W, and latitudes ranging from 30°N to 4°S.

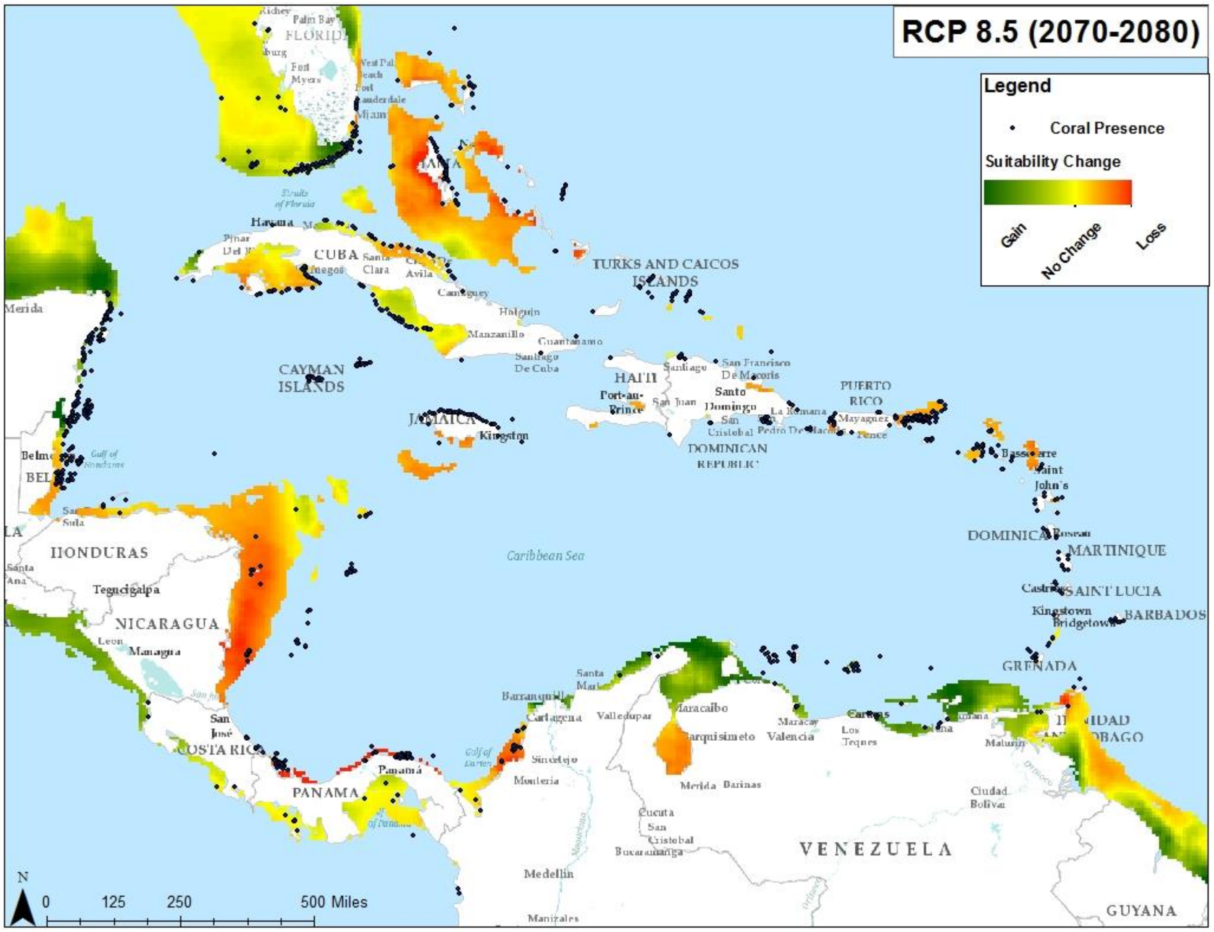

Figure 10. Difference in coral reef suitable habitat in the future (2070-2080) under RCP 8.5 compared to present conditions for *Acroporidae, Montastraeidae, Merulinidae*, and *Poritidae* coral families. Color scale indicates a gain or loss in the probability of suitable habitat for coral reefs. Green areas indicate an increase in the probability of high suitability, yellow areas suggest little to no change in the probability of medium suitability, and red areas indicate a decrease in the probability of low suitability. This region of study is bounded by longitudes ranging from 103°W to 50°W, and latitudes ranging from 30°N to 4°S.

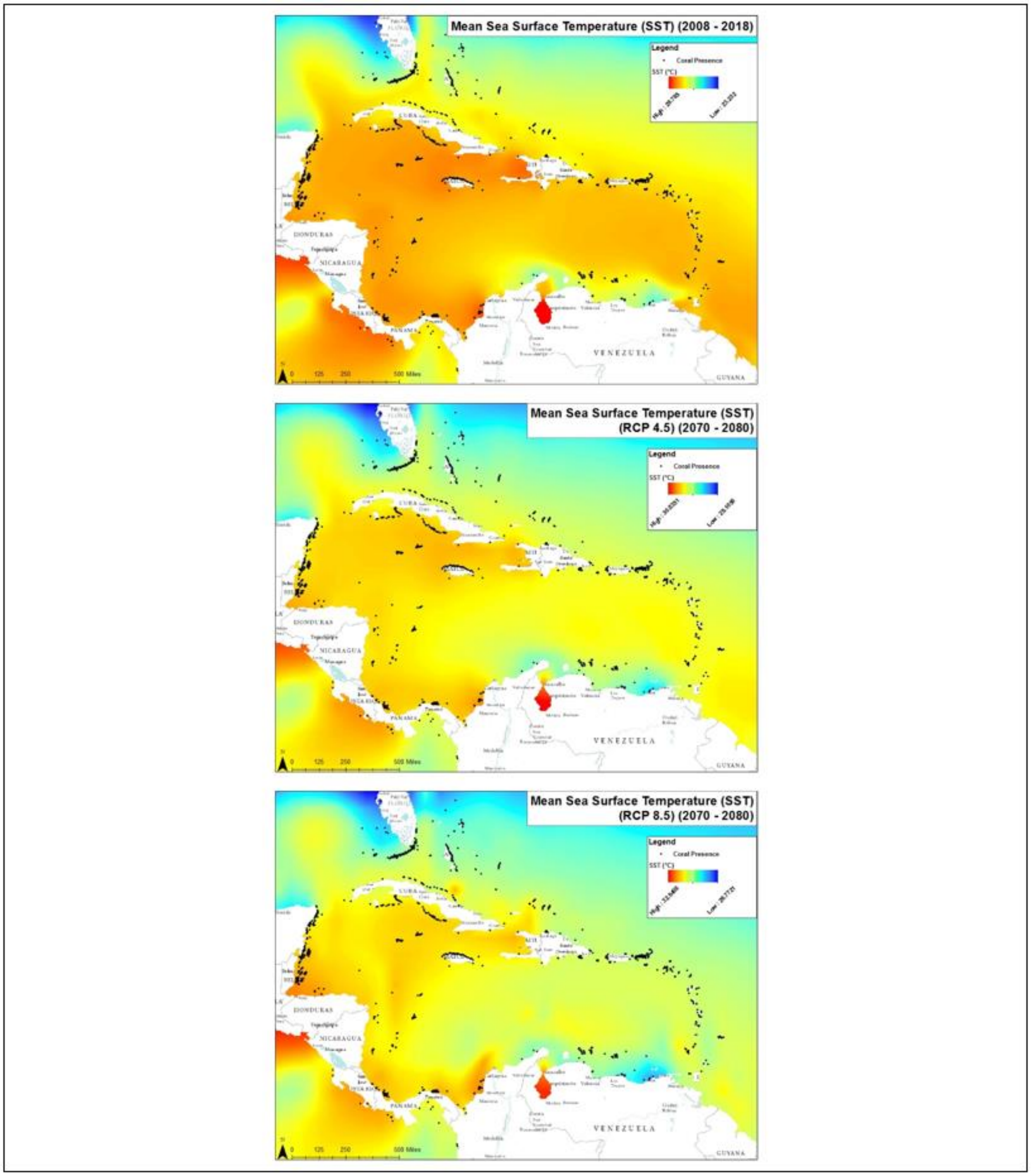

Figure 11. Three charts showing observed and modeled mean sea surface temperatures (SST) under RCP 4.5 and RCP 8.5. Color scale indicates SST (°C), red areas indicate high temperatures and blue areas indicate low temperatures. Charts from top to bottom display observed SST (2008-2018), modeled SST to 2080 under RCP 4.5, and modeled SST to 2080 under RCP 8.5.

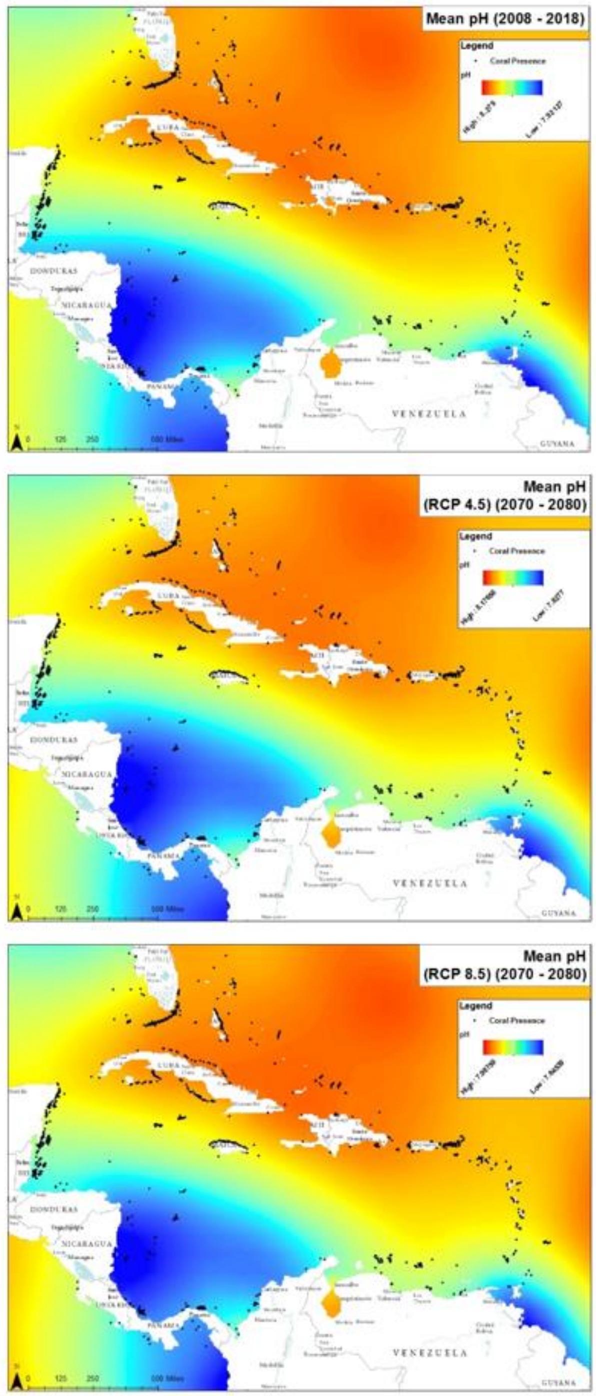

Figure 12. Three charts showing observed and modeled mean pH levels under RCP 4.5 and RCP 8.5. Color scale indicates pH levels, red areas indicate high pH and blue areas indicate low pH. Charts from top to bottom display observed pH (2008-2018), modeled pH to 2080 under RCP 4.5, and modeled pH to 2080 under RCP 8.5.

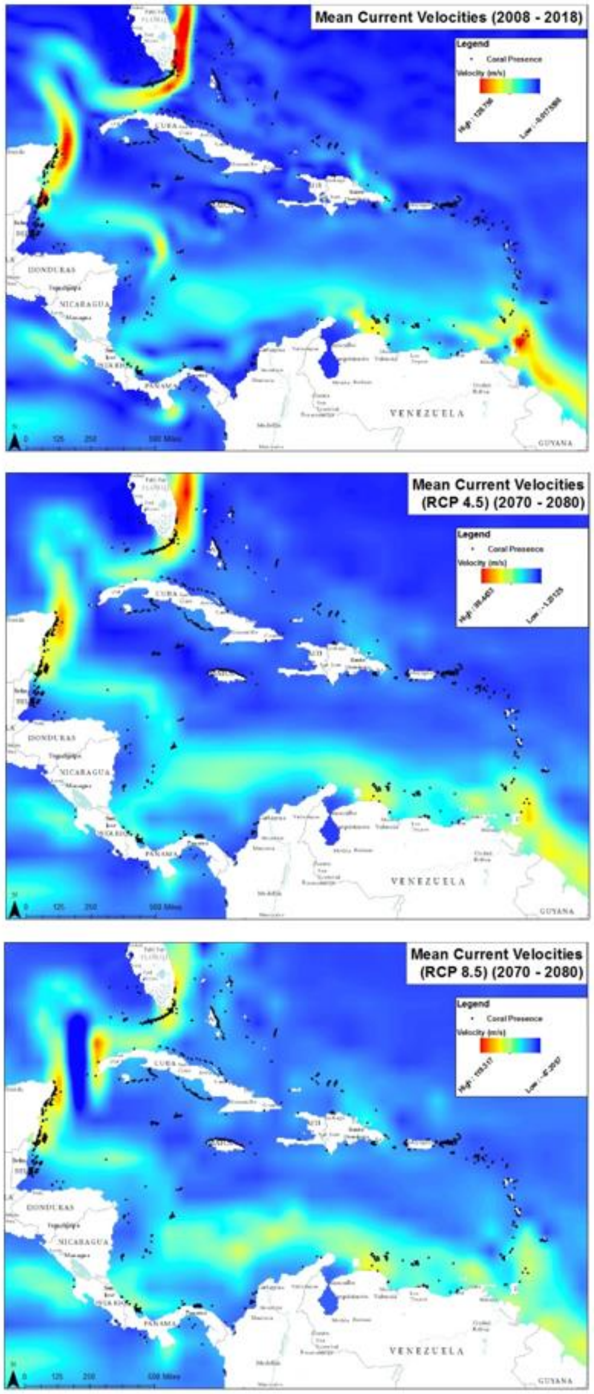

Figure 13. Three charts showing observed and modeled mean current speeds under RCP 4.5 and RCP 8.5. Color scale indicates current speeds (m/s), red areas indicate high speeds and blue areas indicate low speeds. Charts from top to bottom display observed current speeds (2008-2018), modeled current speeds to 2080 under RCP 4.5, and modeled current speeds to 2080 under RCP 8.5.

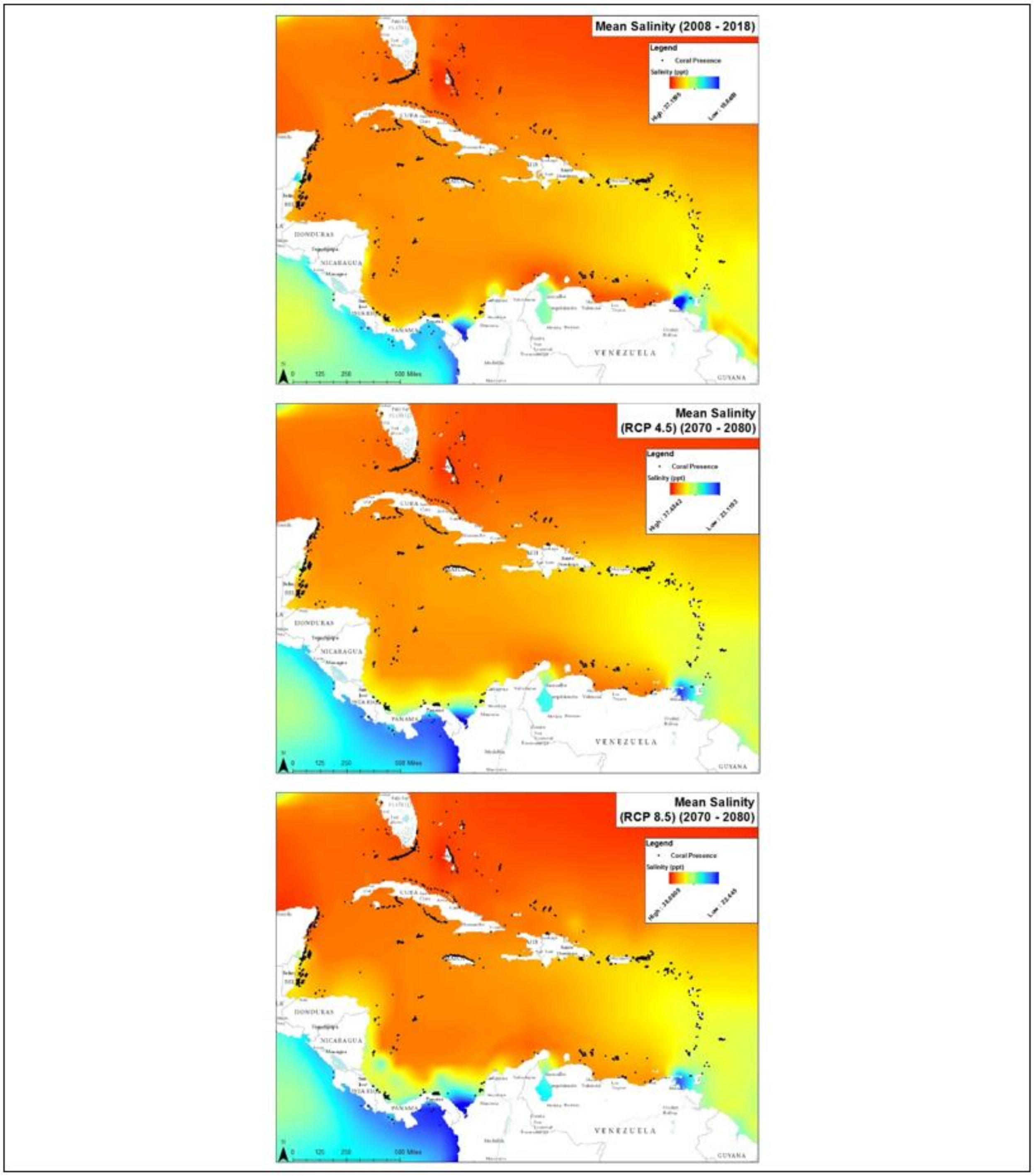

Figure 14. Three charts showing observed and modeled mean salinity concentrations under RCP 4.5 and RCP 8.5. Color scale indicates salinities (ppt), red areas indicate high salinity concentrations and blue areas indicate low salinity concentrations. Charts from top to bottom display observed salinities (2008-2018), modeled salinities to 2080 under RCP 4.5, and modeled salinities to 2080 under RCP 8.5.

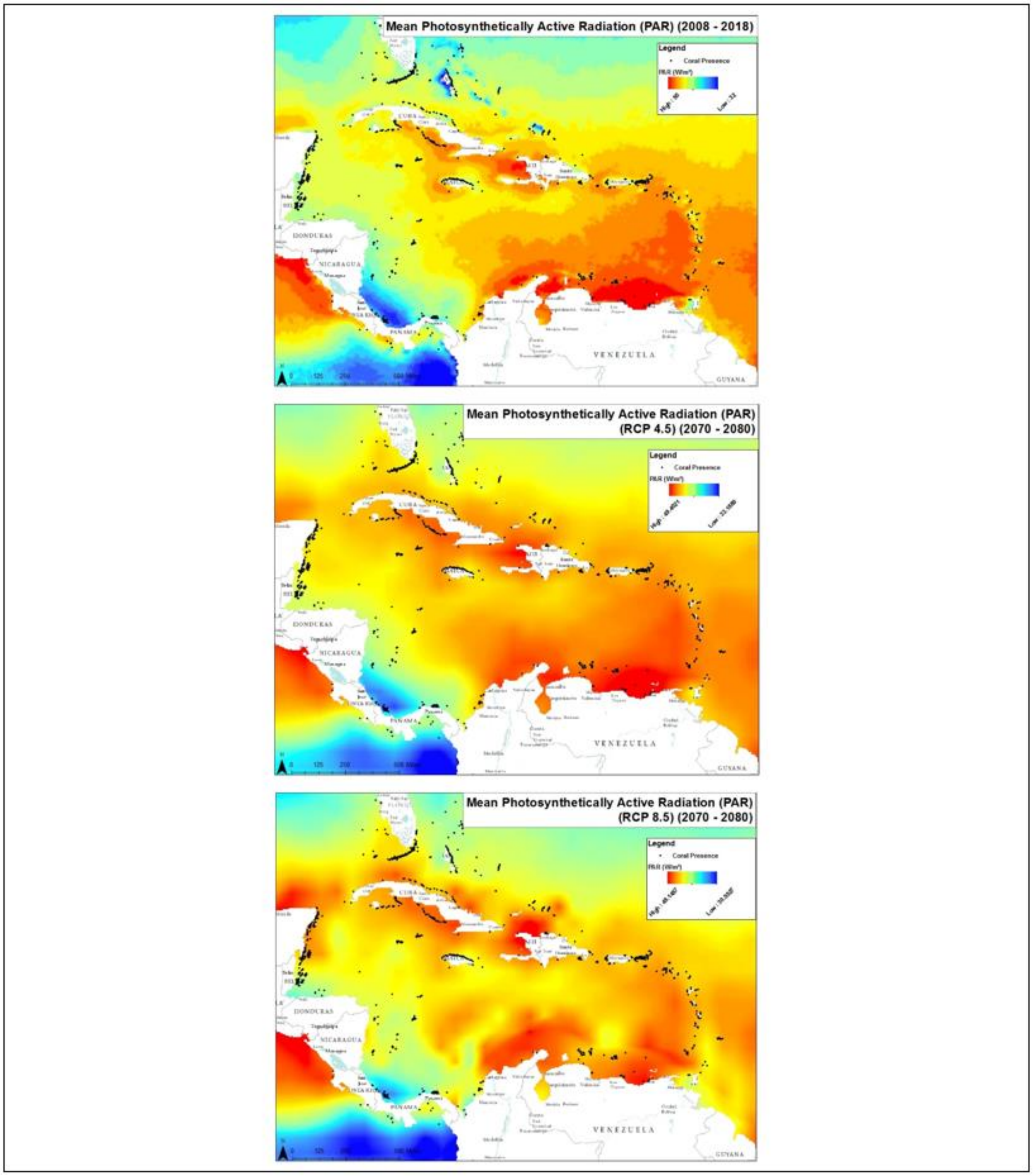

Figure 15. Three charts showing observed and modeled mean photosynthetically active radiation (PAR) levels under RCP 4.5 and RCP 8.5. Color scale indicates PAR (W/m²), red areas indicate high levels and blue areas indicate low levels. Charts from top to bottom display observed PAR (2008-2018), modeled PAR to 2080 under RCP 4.5, and modeled PAR to 2080 under RCP 8.5.