FLORIDA INTERNATIONAL UNIVERSITY

Miami, Florida

COST BENEFIT ANALYSIS OF RESTOCKING THE THREATENED CARIBBEAN STAGHORN CORAL ON THE FLORIDA REEF TRACT

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To: Dean Michael R. Heithaus
College of Arts, Sciences, and Education

This dissertation, written by Kevin Cavasos, and entitled Cost Benefit Analysis of Restocking the Threatened Caribbean Staghorn Coral on the Florida Reef Tract, having been approved in respect to style and intellectual content, is referred to you for judgment.

We have read this dissertation and recommend that it be approved.

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ABSTRACT OF THE DISSERTATION

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Professor Mahadev Bhat, Major Professor

Once a dominant structure building coral on shallow water reefs throughout the Caribbean and western Atlantic, staghorn coral (*Acropora cervicornis*) has experienced precipitous regional declines in abundance since the 1970s, the result of a multitude of interlinked natural and human-induced stressors. To mitigate declining trends and support the recovery of wild staghorn populations, a restocking program has been initiated to transplant tens of thousands of nursery-reared staghorn colonies annually onto reefs off SE Florida and throughout the Caribbean.

The objective of the present study is to examine the business case for a large-scale staghorn coral restocking program in the Florida Keys considering (1) one of the most important non-market functions of staghorn coral in the Florida Keys, support of commercial reef fish fisheries, and (2) the public's willingness-to-pay (WTP) to restock staghorn coral populations.

We develop a multi-stock fisheries bioeconomic model that incorporates the empirical relationship between staghorn coral abundance and commercially important reef fish carrying capacity on the FRT to quantify changes in optimal equilibrium reef fish stocks, harvest, and fishery profit from restocking staghorn coral populations under alternative fishery management regimes.

Using stated preference data elicited through a household survey of residents of the SE US, we estimate the public's willingness-to-pay (WTP) for enhanced staghorn abundance and ecosystem health on the Florida Reef Tract. We integrate psychometric measures characterizing the public's attitudes toward risk into an economic discrete choice model to examine the impact of individual risk characteristics on household WTP. Results of the survey confirm the public assigns substantial value to the recovery of staghorn coral populations and improved coastal ecosystem health on the Florida Reef Tract. Respondent WTP was strongly dependent on individual perceptions of the anthropogenic risks facing staghorn corals and local coral reef ecosystems. Bioeconomic model results suggest staghorn restocking could play an important role in the recovery of locally exploited reef fish stocks, although the incremental economic contribution to the fishery is substantially less than estimated annual WTP values. Benefit cost ratios range from .66 to 36.84 depending on the population of beneficiaries considered.

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1.0 Introduction and background

1.1 Introduction

Coral reefs are some of the most productive and diverse ecosystems on earth, possessing extraordinary biological richness and providing food and resources to more than 500 million people in over 100 countries (Wilkinson, 2008). Estimates of coral reef cover range from only 0.1–0.5% of the ocean floor (Smith, 1978; Copper, 1994; Spalding and Grenfell, 1997), yet nearly one-third of the world's marine fish species are found on coral reefs (McAllister, 1991). Coral reefs can be found in shallow lagoons (platform reefs), along shorelines (fringing reefs), offshore (barrier reefs), and as isolated shallow areas in the open ocean (atolls), generally in areas of warm, clear, shallow, nutrient poor waters (Moberg and Folke, 1999).

Healthy coral reef ecosystems provide a multitude of goods and services of value to people. Coral reef related fisheries account for an estimated 10-13% of the global fisheries catch (Munro and Williams, 1985), providing a variety of seafood products such as mussels, crustaceans, sea cucumbers and seaweeds (e.g., Craik *et al.*, 1990; Birkeland, 1997) to millions of people. Pharmaceuticals and medical products have been derived from corals and reef dwelling organisms that include potential cures for cancer, arthritis, viruses, and other diseases (e.g., Sorokin, 1993; Carte´, 1996; Birkeland, 1997). High numbers and diversity of marine species are drawn to the complex structure of coral reefs, supporting fisheries, tourism, recreation, educational and spiritual experiences

(Wilkinson, 2008; Principe *et al.*, 2012). The physical structure of coral reefs also provides physical coastal protection that can help mitigate coastal flooding, property damage and loss of life associated with large tropical storms (Sudmeier-Rieux *et al.*, 2006; Goreau *et al.*, 2012; Guannel *et al.*, 2016).

The world's coral reefs are in peril, their natural resilience compromised by the cumulative effects of over-exploitation, pollution, habitat destruction, invasive species, disease, bleaching and global climate change (NMFS, 2015). In 2006, staghorn coral (*Acropora cervicornis*) and elkhorn corals (*A. palmata*) became the first marine invertebrates to be classified as 'threatened' under the US Endangered Species Act (NMFS, 2006). Twenty additional species of corals have been added to the list since that time, five of which occur in the Caribbean and 15 in the Indo-Pacific. More than half of the world's reefs are presently under medium or high risk of degradation (Burke *et al.*, 2011), and research increasingly suggests that unavoidable climate change impacts makes corals' global extinction possible within decades.

Staghorn is a stony coral characterized by straight or slightly curved antler-like, cylindrical branches ranging from a few centimeters to over two meters in length (Gladfelter, 1983; Tunnicliffe, 1983). Studies of fossilized corals indicate the shallow fore-reef zones of the Caribbean region were once dominated by staghorn thickets (Pandolfi & Jackson, 2006; Precht & Aronson, 2006). The dominance of asexual reproduction through fragmentation in staghorn corals and limited larval dispersal have led to diminished effective population sizes and low genetic variation in regional populations, resulting in increased risk of disease (Bak, 1983). Since the 1970s, declines in the abundance of staghorn corals off Florida have been estimated as high as 97% in some locations, primarily the result of white-band disease (Aronson and Precht, 2001), but linked to many inter-related natural and human induced stressors (NMFS, 2015).

Impediments to the recovery of the species regionally include disease, increasing temperature, depensatory population effects, loss of recruitment habitat, sedimentation, natural and human caused abrasion and breakage, predation, inadequacy of existing regulatory mechanisms, ocean acidification, and nutrients and contaminants (Aronson and Precht, 2001; Bruckner, 2002; Hughes *et al.*, 2003; NMFS, 2015). The widespread loss of the three-dimensional branching structure of staghorn corals from regional waters has dramatically reduced essential habitat and feeding, breeding, and spawning grounds for many economically important fish and invertebrates, likely impacting biodiversity and fisheries productivity and value.

Research suggests restocking staghorn colonies on denuded reefs may support the long-term recovery of wild populations and their genetic diversity (Lirman *et al.*, 2014). A common propagation and restoration method, “coral gardening”, entails extracting small amounts of tissue and skeleton from healthy wild coral colonies to propagate nursery stocks (in situ or ex situ) from which fragments can be pruned and transplanted to degraded reefs (Rinkevich, 1995, Bowden-Kerby, 2001; Epstein *et al.*, 2001; Shafir and Rinkevich 2008; Shaish *et al.*, 2008). Rapid growth rates and ability to reproduce through asexual propagation make staghorn coral well-suited for restocking projects (Highsmith, 1982; Lirman, 2010; NOAA, 2012). Multiple staghorn restocking projects have experienced high levels of success in the Caribbean and Florida Keys since the early 2000s (Schopmeyer *et al*, 2017).

This research attempts to examine the economic efficiency of restocking denuded reefs with nursery-reared staghorn colonies by quantifying two of the most important non-market economic values impacted by active coral reef restoration: support of commercial reef fish fisheries and support of non-consumptive recreational coral reef uses like diving and snorkeling. In Chapter One of this paper we use an existing bioeconomic fishery model (Conrad, 1999), parameterized using locally collected fishery data (Miller and Huntington, 2015; SEFSC, 2016), to attempt to quantify the potential impact to the value of local commercial reef fish fisheries from efforts to restock staghorn coral populations. In Chapter Two we use two stated preference (SP) techniques to examine the public's willingness-to-pay to support staghorn coral populations off SE Florida. In Chapter Three we synthesize the findings from Chapters One and Two and, incorporating outplanting and monitoring cost data, derive the discounted net present value of the fishery and benefit-cost ratio under several hypothetical large-scale staghorn restocking scenarios.

1.2 Coastal resource valuation

Consideration of the economic values of goods and services flowing from marine resources is essential to decisions regarding their efficient use and allocation. Recognizing the universal importance of coral reefs, economists have spent several decades working to improve the reliability of estimates of their values. Early coral reef valuation studies tended to focus on direct-use values, like recreational snorkeling, diving, and fishing (e.g., Hundloe, 1990; Leeworthy, 1991; Leeworthy and Bowker, 1997; Johns *et al.*, 2001; Cesar *et al.*, 2002; Brander, 2006); Recent studies have attempted to estimate the changes in direct-use values of coral reefs or recreational destinations associated with proposed management decisions or policy changes (e.g., Cesar and Chong, 2002; Bhat, 2003; Bishop *et al.*, 2011). Published estimates of the most important direct-use values exist for coral reef ecosystems in all US jurisdictions (Brander and van Beukering, 2013), however, the value of contributions from indirect uses, like essential habitat for commercially important fish stocks, are less common in the literature. Numerous studies have used mathematical simulation models to examine the bioeconomics of habitats supporting coastal fisheries (e.g., Lynne, *et al.*, 1981; Bell, 1989; Bell, 1996; Barbier and Strand, 1997; Sathirathai, 1997; Barbier, 2000; Foley, *et al.*, 2012). By quantifying biophysical connections between habitat quantity and/or quality and fishery productivity, these studies generally attempt to estimate changes in equilibrium stocks, effort, yield and /or profits under selected property rights regime(s), typically "open access", but commonly maximum sustainable yield (MSY) or maximum economic yield (MEY). Partitioning a simulated optimally managed fishery into a marine reserve (MR) and fishing grounds, Conrad (1999) compared optimal stocks, harvest and profits under various MR and fishing ground configurations. Lynne (1981) examined the role of marshlands of South Florida in supporting Gulf Coast fisheries by estimating the relationship between harvest, fishing effort, and marsh area. Bell (1996) estimated a fisheries production function to quantify the incremental value of saltwater marsh on recreational fish catch and consumer surplus. Findings suggest when considering the value of wetlands in supporting recreational fisheries, a state policy of purchasing and preserving coastal wetlands from development may be the most economically efficient. Modifying an open-access fisheries model to account for the effect of changes in mangrove area on equilibrium harvest and effort, Barbier and Strand (1997) demonstrate the detrimental effect of mangrove loss on the shrimp fishery of Campeche State, Mexico. Similarly, Sathirathai and Barbier (1997) used the Ellis-Fisher-Freeman model to estimate welfare effects of changes in mangrove area on Gulf of Thailand fisheries under open-access and managed fishery conditions.

The purpose of our study is to quantify the indirect economic and ecological benefits from coral reef restoration on the Florida Reef Tract (FRT). We develop a model that establishes a value for one of the non-market functions of staghorn corals, namely support of commercial fisheries, by exploring the empirical relationship between staghorn coral (*Acropora cervicornis*) abundance and commercially important reef fish carrying capacity on the FRT. This technique is consistent with previous efforts to examine the non-market benefits of natural systems (e.g., Lynne *et al.*, 1981; Ellis *et al.*, 1987; Barbier and Strand,1997; Loomis, 1998). We first simulate growth of coral colonies transplanted onto denuded reefs, then embed the abundance of outplanted coral as an environmental input into a multi-stock fishery bioeconomic model (Conrad, 1999) to enable comparison of changes in optimal equilibrium stocks, harvests, and fishery value from restocking under open-access and managed fishery regimes. To our knowledge, our study is the first empirical application of Conrad's (1999) model using fishery specific parameters and data and contributes to the existing ecological-economic literature by creating a framework for evaluating the commercial fishery benefits from restocking and protecting staghorn corals.

1.3 Study area

The Florida Reef Tract (FRT) reaches approximately 220 miles southwest from Soldier Key off Miami to the Tortugas Banks in the Gulf of Mexico. About two-thirds of the FRT lies inside Biscayne National Park and the Florida Keys National Marine Sanctuary (FKNMS), a 2,900-square nautical mile (NM^2) marine protected area (MPA) that surrounds the Florida Keys. Proximity to the Miami metropolitan area and Florida Keys has subjected the reef ecosystem to decades of intense human use. Bruckner (2002) found mean staghorn coverage on the FRT to be 0.049% with little variation among the eight habitat types surveyed; Twenty- three of 35 species of groupers, snappers, hogfish, and grunts have been chronically over-fished since the 1970s according to National Marine Fishery Service (NMFS) standards (Ault, 1998). Partially in response to fishing pressure, 18 sanctuary preservation areas (SPA), totaling 1.45 NM^2, were established in 1997 in the FKNMS. The Tortugas Ecological Reserve (TER) was created as part of the FKNMS in 2001 to protect coral reef ecosystems and support reef fisheries. The TER protects 150 NM^2 prohibiting anchoring, fishing and other extractive activities bringing the aggregate area closed to all fishing in the Keys and Tortugas region to about 200NM^2, 150NM^2 in the TER, 35 NM^2 in the Research Natural Area in Dry Tortugas National Park, 9 NM^2 in Western Sambo Ecological Reserve, and 1.45 NM^S in the SPAs. Populations of several species of exploited reef fish, including black grouper, red grouper, and mutton snapper, have experienced dramatic increases in abundance since the TER was designated in 2001 (Ault *et al*., 1999), however, staghorn coral populations have shown little to no sign of natural recovery regionally.

Staghorn coral, which can form large thickets two to three meters in height and 30 meters long (NMFS, 2015), was once a dominant coral in terms of structure accretion on shallow reef slope and fore reef environments in the Caribbean region. Staghorn historically occurred in SE Florida on the outer reef (Goldberg, 1973), spur-and-groove bank and transitional reefs (Jaap, 1984, Wheaton and Jaap, 1988), and consolidated hardbottom (Davis, 1982); Today, staghorn corals on the FRT exist primarily as isolated colonies or small thickets on shallow patch reefs (Miller *et al.*, 2008). In 2006, staghorn coral became one of two marine invertebrates classified as 'threatened' on the US Endangered Species (ES) List (NMFS, 2006). Strategies identified to rebuild wild populations include restocking denuded reefs on the FRT with nursery-reared staghorn colonies and designation of "no-take" marine reserves to support outplanted colonies and restocked reefs (NMFS, 2015).

Changes in the structure and function of the coral reef ecosystems affect the life cycle and population dynamics of commercially harvestable reef fish species (Syms and Jones, 2000) and, thus, fishery productivity and value. Promoted as the "Fishing Capital of the World,", Florida is dependent on the health of its coastal resources to support sectors of the state's economy reliant on tourism and outdoor recreation. In 2012, the commercial fishing industry of East Florida supported over 82,000 jobs with landings of almost 13 million kg while the recreational fishing industry supported over 34,000 jobs and sales of over $4.0 billion (NMFS, 2014). Over the same period as the precipitous decline in staghorn coral populations in SE Florida, mean annual commercial fishery landings off the east coast of Florida have fallen from over 37 million kg in 1980 to under 13 million kg for the period 2010-2016 (https://www.st.nmfs.noaa.gov). In our paper, using extensive field data collected in the Dry Tortugas National Park (DTNP) (Miller and Huntington, 2015) by the National Oceanic and Atmospheric Administration (NOAA), we try to quantify the relationship between staghorn coral coverage and the abundance of commercially important fish. With this empirical test, we aim to investigate whether staghorn restocking can benefit the commercial reef fishery and its economic bottom line in the Florida Keys.

Since the 1980s, federal agencies have been required to prepare analyses examining the economic efficiency of major policy decisions such as marine regulations and restorations. Currently, no published studies examine the long-range economic viability of restocking and protecting staghorn coral populations on the FRT. Our research estimates the *ex-ante* commercial reef fish fishery impacts of restocking staghorn corals under alternative outplanting intensities and fishery management / property rights regimes. Establishing a value for one of the primary non-market functions of restocked staghorn populations can inform cost-benefit analyses and support efforts by policy and decision makers to compare the potential benefits of alternative staghorn restoration projects and protection regimes, prioritize restoration and protection programs or projects, and maximize the ecological benefits per dollar spent. While reef protection supports a host of other non-market and market benefits (Moberg and Folke, 1999), valuing every one of them is beyond the scope of our paper.

Subsequent sections develop the theoretical and empirical methods used to examine the relationship between staghorn coral abundance and coral reef fish biomass. We first assume the fishery is managed for MEY, one-third (100 hectares (ha)) of which is restocked with staghorn corals. Next, we examine optimal equilibrium conditions under a marine reserve – fishing grounds configuration whereby the fishery is partitioned, and restocked reefs occupying one-third of the fishery are closed to consumptive uses. The biophysical effects of improved habitat from restocking enter our model through the carrying capacity in the reef fish stock growth function. The fishery impact of protecting restocked reefs enters through the intrinsic growth rate of the stock. We derive the stock and harvest levels achieving the optimal equilibrium of the fishery as well as the comparative static effects of restocking and protecting staghorn corals. We conclude by discussing the management implications of our findings, which we believe are relevant to economic analyses of current restocking efforts on the FRT as well as staghorn coral restoration efforts elsewhere.

2.0 Methodology

2.1 Bioeconomic model of coral-fishery linkages

Bioeconomic models generally integrate biological and economic factors to examine the potential impacts of management actions or variations in ecosystem inputs on the flow of goods and services supported by natural systems (Hanley and Barbier, 2009). Bioeconomic models have been used to examine the linkages between coastal fisheries production and habitats like marshlands, mangroves, and seagrass meadows (Lynne, 1981; Bell, 1989; Barbier and Strand, 1997; Bell, 1997; Kahui, Armstrong, and Vondolia, 2016). Conrad (1999) developed deterministic and stochastic models to examine optimal biomass levels, harvest rate, and fishery value under fishery management / property rights regimes. We modify Conrad's (1999) model to account for the effect of staghorn coral coverage on commercially harvestable reef fish biomass and productivity and quantify changes in the optimal equilibrium commercial reef fish stocks, harvest rate and profit from restocking and protecting staghorn coral populations. Because stocks of the most commercially harvested reef fish in Florida are managed, we first examine equilibrium conditions characterizing maximum economic yield (MEY) and the stocks which maximize net economic benefits to society, rather than that of an open access fishery. Following Barbier and Strand (1997), we simulate multiple scenarios to examine the comparative static effects of changes in staghorn area on equilibrium conditions. This approach is the first empirical and management application of multi-stock bioeconomic fisheries model linked with staghorn restocking on the FRT, and allows evaluation of alternative combinations of management actions, namely restoration and marine protection versus no action.

To estimate comparative static effects of restocking and protecting staghorn corals on equilibrium conditions, we examine changes in optimal commercial reef fish stocks and harvest from restocking staghorn coral under two fishery management regimes: 1) optimally managed fishery with no marine reserve, and; 2) fishery with marine reserve, i.e., fishing grounds, a portion of which is managed for MEY and the other portion as a no-take marine reserve. We first examine the model of the optimally managed fishery.

We use a dataset of reef fish and staghorn colony measures and abundance collected between 2012-2014 using underwater visual surveys (n=65 transects) in the Dry Tortugas National Park (Miller and Huntington, 2015), an area of relatively rich coral reef ecosystems, to estimate staghorn coverage, reef fish biomass, and quantify the relationship between the two. Using an observational dataset of reef fish measures inside and outside of no-take marine reserves in the FKNMS (SEFSC, 2016), we estimate mean reef fish biomass in the study area to be 213.98 kg ha^{-1} and 134.51 kg ha^{-1}, respectively. We use these initial biomass values, with the estimated relationship between staghorn coverage and reef fish biomass, to estimate increases in biomass associated with enhanced staghorn cover from restocking.

Table 1. Florida Keys commercial reef fish biomass and DRTO staghorn cover and commercial reef fish biomass

Statistic	FKNMS protected areas (n=202 transects)	FKNMS unprotected areas (n=595 transects)	Dry Tortugas National Park (DRTO) (n=65 transects)	DRTO staghorn coral cover (%) (n=65 transects)
Mean	335.79	230.63	1180.51	18.83
Median	213.98	134.51	621.00	7.23
Max.	2452.35	5376.69	6202.39	77.66
95th pct.	1220.22	730.75	5142.13	54.31
Std. dev.	404.53	401.50	1636.46	20.16

2.2 Optimally managed single stock fishery

Before examining the fishery partitioned into a marine reserve and fishing grounds, we first introduce a model of an optimally managed single-stock fishery whereby stocks and harvest are managed to maximize the economic yield of the fishery. Following Conrad (1999), biomass of commercially harvestable reef fish and harvest rate at instant t are denoted $X = X(t)$ and $Y = Y(t)$, respectively. Suppose that $\pi(X, Y)$ is the annual net income from the commercial harvest (Y), which increases at a decreasing rate with respect to stock and harvest. The annual growth of stock follows the equation of motion, $dX/dt = \dot{X} = F(X) - Y,$ where $F(X)$ is a strictly concave net growth function. Applying the Maximum Principle, the stock size at the steady-state optimum must satisfy

$F'(X) + \pi_X/\pi_Y = \delta$ and $Y = F(X)$, where $\pi_x = \partial\pi(\cdot)/\partial X$, $\pi_Y = \partial\pi(\cdot)/\partial Y$, and δ is the discount rate (Clark, 1990). The steady state bioeconomic optimum is denoted (X^*, Y^*). We model the optimally managed fishery such that $X = X^*$and $Y^* = F(X^*)$.

If $\pi(X, Y) = (p - c/X)Y$ and $F(X) = rX(1 - X/K)$, where $p > 0$ is the unit price for fish on the dock, $c > 0$ is a cost parameter, $r > 0$ is the intrinsic fish stock growth rate the and $K > 0$ its environmental carrying capacity, then the optimal equilibrium biomass level is

$$X^* = \left[\frac{K}{4}\right]\left[\left(\frac{c}{pK} + 1 - \frac{\delta}{r}\right) + \sqrt{\left(\frac{c}{pK} + 1 - \frac{\delta}{r}\right)^2 + \frac{8c\delta}{pKr}}\right] \qquad [1]$$

and $Y^* = rX^*(1 - X^*/K)$. The net present value (NPV) at the bioeconomic optimum is

$$V(X^*) = (p - c/X^*)rX^*(1 - X^*/K)/\delta. \qquad [2]$$

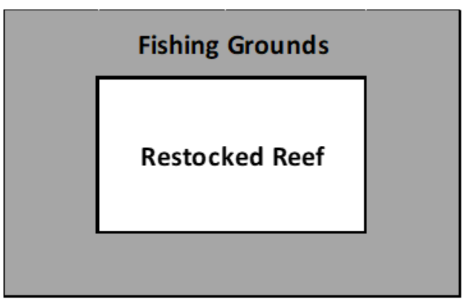

Figure 1. Marine reserve - grounds configuration

Now, we turn to two-stock model of reef-dependent commercial fishing whereby the economic yield of the fishery is maximized subject to partitioning the fishery into a marine reserve, which receives staghorn outplants, and a fishing ground which receives no staghorn outplants. Figure 1 represents our simulated fishery containing a coral reef restocked with nursery-reared staghorn corals. The purpose of the following model is to link fishery productivity on the fishing grounds to the restocking and protection of the reef. To enable comparison of equilibrium conditions between the two management regimes, we use a dynamic model of optimal fishery harvesting (Conrad, 1999). Formally, a management agency's objective is to maximize the present value net benefit of

$$Z(K_1, K_2) = \int_0^T \pi(X_1, Y) e^{-\delta t} dt, \qquad [3]$$

where K_1 and K_2 are the carrying capacities for commercially harvestable fish on the fishing grounds and in the marine reserve containing the restocked reef, X_1 and X_2 are the stocks on the fishing grounds and the marine reserves, respectively, and Y is harvest from the grounds in period t and is subject to a finite upper bound, Y_{MAX}, and a lower bound of zero. The underlying growth dynamics of the fish stock on the fishing grounds is assumed to follow:

$$\dot{X}_1 = F_1(X_1) + s\left(\frac{X_2}{K_2} - \frac{X_1}{K_1}\right) - Y \qquad [4]$$

where $F_1(X_1)$ is a density dependent logistic growth function, $s > 0$ is a migration coefficient, and X_2 is the stock of harvestable fish in the reserve; and growth in the reserve:

$$\dot{X}_2 = F_2(X_2) - s\left(\frac{X_2}{K_2} - \frac{X_1}{K_1}\right), X_1(0) \text{ and } X_2(0) \text{ given } Y_{MAX} \geq Y \geq 0 \qquad [5]$$

The population dynamics of commercially harvestable fish, $F_i(X_i)$, we simulate using the logistic function:

$$rX(1 - X/K) \qquad [6]$$

where $r > 0$ is the intrinsic rate of growth for the fish stock. Because the fishing grounds receives no coral outplants, K_1 remains fixed over time; K_2 increases subject to the growth of outplanted corals. We assume restocking and protecting the reef results in migration of fish to the grounds from the reserve, expressed as a constant proportion of the difference in the pressures on the respective populations, $s\left(\frac{X_2}{K_2}-\frac{X_1}{K_1}\right)$, where $s>0$.

We compute the optimal stocks and harvesting paths to the above problem by solving the following current-value Hamiltonian (Conrad, 1999):

$$\widetilde{H}=\pi(X_1,Y)+\mu_1\left[F_1(X_1)+s\left(\frac{X_2}{K_2}-\frac{X_1}{K_1}\right)-Y\right]+\mu_2\left[F_2(X_2)-s\left(\frac{X_2}{K_2}-\frac{X_1}{K_1}\right)\right], \quad [7]$$

where μ_1 and μ_2 are the current value shadow prices for reef fish on the grounds and in the reserve, respectively. Because we assume optimal management (and, thus, optimal fish stocks) prior to restocking, equilibrium harvest at $t=0$ is equivalent to the sum of net reef fish stock growth on the fishing grounds and in the sanctuary

$$Y^*=r_1X_1(1-X_1/K_1)+r_2X_2(1-X_2/K_2) \quad [8]$$

The optimal equilibrium fish stock on the grounds and in the reserve, (X_1^*,X_2^*), must also satisfy

$$F_1'(X_1)+\frac{c[F_1(X_1)+F_2(X_2)]}{X_1^2(p-c/X_1)}+\left[\frac{S^2}{K_1K_2[\delta-F_2'(X_2)+s/K_2]}\right]-\frac{s}{K_1}-\delta=0, \quad [9]$$

equivalent to requiring that the reef fish stock earn a rate of return commensurate with that which could be earned elsewhere in the economy, δ (Conrad, 1999). Using *ex-ante* estimates of outplanted staghorn coverage, reef fish fishery carrying capacity, market fish prices, and parameters derived from peer reviewed literature, our model enables characterization of the linkages between staghorn outplanting and protection, and commercial reef fish stocks and optimal sustainable harvest.

2.3 Derivation of model parameters

The cost parameter, c, was derived from a National Marine Fisheries Service (NMFS) survey of commercial vessels in the Gulf of Mexico (NMFS, 2016). From the NMFS fisher survey dataset, we calculate total variable and fixed costs to be 60.3% of revenue (Table 2). This is equivalent to the expression $\frac{c}{X} = .603 * p,$ where c/X is the unit cost of harvest and p denotes market price. Rearranging terms, we solve for c, total cost of harvest: $c = .603 * p * X$. Using the market price of \$5.87 kg^{-1} and median reef fish abundance in unprotected areas estimated from the SEFSC (2016) datasets (134.51 kg ha^{-1}) multiplied by the size of the grounds (200 ha), we calculate total harvests costs to be \$95,222. We assume commercial vessels are owner operated, therefore, captain pay is embedded in boat profit rather than presented as a percentage of total costs.

Table 2. Commercial fishing costs

Expense	% of revenue
Fuel	11.8
Bait	8.30
Ice	2.40
Groceries	3.50
Miscellaneous	2.50
Tackle	2.80
Captain Pay	0.00
Crew Pay	19.1
Overhead (assumed)	10.0
Total variable and fixed costs	60.3

Source: NMFS (2016)

Per kilogram fish price on the dock, p, was taken from NMFS landing data collected from 2012-2014. The rate of discount of 4% is the mean 10-year US Treasury note yield since 1997 (3.9%), rounded up to the nearest whole number; discount rates of 2%-6% are commonly used in the literature. The fish stock growth rates on the grounds and in the reserve, r_1 and r_2, respectively, in the reserve - grounds configuration were taken from www.fishbase.org (Froese and Pauly, 2018). In the bioeconomic literature, biophysical effects of habitat change typically enter the model through the stock growth function (Barbier, 2000; Foley, *et al*., 2012). To account for improved habitat and increased fishery productivity from protecting restocked colonies, the reef fish stock growth function in our model is greater inside the reserve than outside the reserve in the marine reserve – grounds configuration. The migration coefficient, $s > 0$, presumes fish move from the reserve to the grounds in search of more plentiful food or less congested habitat and is estimated to be 10% of the carrying capacity of the fishery (Conrad, 1999). Because movement of fish in and out of marine reserves is difficult to track reliably and limited data exists, estimation of spillover effects is challenging. Our estimate follows Conrad's (1999) diffusion coefficient of approximately 10% of the carrying capacity of the fishery, however, ours is an educated guess and may under or over represent the actual diffusion of fish from the marine reserve onto the grounds.

Table 3. Bioeconomic model parameters: optimally managed fishery

		Annual Outplants		
Parameters	Description	50,000	40,000	30,000
c	Annual harvest cost	190,452	152,360	114,270
δ	Discount Rate	0.04	0.04	0.04
p	Unit price fish at dock ($/kg)	5.87	5.87	5.87
r	Intrinsic growth rate on grounds	0.20	0.20	0.20
K	Fishery carrying capacity	64,194	51,355	38,516
	Fishery size (ha)	300	240	180

Table 4. Reef fish species evaluated

Common Name	Scientific Name
White Grunt	*Haemulon plumierii*
Bluestripe Grunt	*Haemulon sciurus*
Red Grouper	*Epinephelus morio*
Black Grouper	*Mycteroperca bonaci*
Yellowtail	*Ocyurus chrysurus*
Gray Snapper	*Lutjanus griseus*
Mutton Snapper	*Lutjanus analis*
Hogfish	*Lachnolaimus*

Table 5. Bioeconomic model parameters: marine reserve-fishing grounds configuration

		Outplant number		
Parameters	Description	50,000	40,000	30,000
c	Annual harvest cost	95,226	76,180	57,135
δ	Discount rate	0.04	0.04	0.04
p	Unit fish price at dock ($/kg)	5.87	5.87	5.87
r_1	Intrinsic growth on grounds	0.20	0.20	0.20
r_2	Intrinsic growth in reserve	0.30	0.30	0.30
K_1 (constant)	Carrying capacity grounds	42,797	34,237	25,678
K_2	Carrying capacity reserve	21,398	17,119	12,839
s	Spillover coefficient	6,419	5,136	3,852
	Grounds size (ha)	200	160	120
	Reserve size (ha)	100	80	60

2.4 Carrying capacity estimation

We use median reef fish density in FRT marine reserves estimated from the SEFSC (2016) dataset to derive fishery carrying capacity (kg ha^{-1}). We converted length – weight observations (n=202 transects) for eight species of commercially harvestable groupers, snappers, and grunts (Table 4) to biomass using the equation: $W = \alpha L^b$ where W is the weight (gm), L is the length to fork (cm), and α and b are parameters estimated by linear regression of logarithmically transformed length-weight data (Bohnsack and

Harper, 1988). Marine reserve and fishing ground carrying capacity at $t = 0$ are calculated as the product of the median biomass from the SEFSC (2016) dataset (213.98 kg ha^{-1}) and the number of hectares in the respective area:

$$Marine\ reserve\ carrying\ capacity = 213\ kg\ ha^{-1} * 100\ ha = 21{,}398\ kg \quad [10]$$

$$Fishing\ grounds\ carrying\ capacity = 213\ kg\ ha^{-1} * 200\ ha = 42{,}796\ kg \quad [11]$$

2.5 Simulating growth of outplanted corals and resulting changes in carrying capacity

At the time of outplanting, simulated colonies are presumed elliptical in shape, 25 cm in length. We simulate changes in coverage of outplanted staghorn colonies following the equation for the area of an ellipse

$$Area = \pi AB \quad [12]$$

where A and B are one-half the length and width of the colonies' major and minor axis, respectively (Kiel, 2014).

From the Miller and Huntington (2015) dataset, the sum of the length, width, and height, or total linear length (TLL), at outplanting was imputed

$$OutplantTLL = \frac{25}{(L/TLL)_{mean}} \quad [13]$$

where 25 is the major axis length and $(L/TLL)_{mean}$ is calculated

$$(L/TLL)_{mean} = \frac{1}{951}\sum_{i=1}^{951} L_i/(L_i + W_i + H_i) \quad [14]$$

where TLL is L_i, W_i, H_i are the length, width, and height, respectively, of the i^{th} colony. Outplant width at $t = 0$ is calculated

$$outplant\ width = W/TLL_{mean} * OutplantTLL \quad [15]$$

where W is colony outplanted colony width, TLL_{mean} is the mean sum of colony length, width, and height calculated from the sample. Simulated outplants are spaced one meter apart (10,000 per hectare (ha)) to increase the potential for cross fertilization of gametes (Johnson *et al*., 2011). We examine treatments of three, four and five hectares and major axis growth rates of three, four, and five cm yr^{-1}. Our baseline results examine treatments of five ha and an annual growth rate of five cm. We cap colony length at 100 cm (at which point colonies in the interior of the treatment will meet and begin to interlock) and cap coverage to 54.31% of the treatment area, which is found to be approximately equal to the 95th percentile coral coverage estimated from the Miller and Huntington (2015) dataset. Simulated outplants in the baseline scenario experience first-and-second year mortality of 15% and 10%, respectively, and none thereafter (Schopmeyer *et al*, 2017). An additional scenario was examined with first-and-second year outplant mortality of 15% and 10%, respectively, and 6% annual die offs in total staghorn area in years 3-20 (Goergen *et al*., 2019).

2.6 Comparative static effects of a change in staghorn area

We quantify the incremental contribution of staghorn coverage to commercial reef fish carrying capacity by regressing the logarithm of reef fish density on staghorn percent coverage estimated from the Miller and Huntington r (2015) dataset (R^2=.7163)

$$lnFish = \beta_0 + \beta_1 lnCoral + \beta_2 GruntDummy + \beta_3 GrouperDummy + \beta_4 SnapperDummy \qquad [16]$$

Dummy variables indicating the dominant fish group in each transect were used to enable examination of individual species effects and characterize the composition of the "average" transect. Reef fish carrying capacity in the restocked area/marine reserve, K_2, in periods 1-20 is calculated

$$Biomass_t = Biomass_{t-1} + (1 + \beta_1) * (Coral\%_t - Coral\%_{t-1}) \qquad [17]$$

where $Biomass_t$ is commercially harvestable reef fish carrying capacity (kg ha^{-1}); Because no restocking takes place on the fishing grounds with the marine reserve – fishing grounds configuration, its carrying capacity, K_1, remains fixed at the $t = 0$ level of 213.98 kg ha^{-1} (equation 11).

Table 6. Regression results for staghorn coverage and reef fish biomass linkages

Variable	Parameter estimate	Standard error	T-statistic	Prob>(T)
Intercept	-6.421	0.8907	-7.2087	1.10e-09
Coral coverage (%)	0.0643	0.0218	2.9502	0.0045
Grunt dummy	9.7037	1.1515	8.4273	9.18e-12
Grouper dummy	12.6313	1.4320	8.8209	1.98e-12
Snapper dummy	11.1385	1.3752	8.0996	3.31e-11

3.0 Results and discussion

Our baseline scenario consists of 50,000 staghorn outplants growing at a rate of 5 cm yr^{-1} with first and second year mortality of 15% and 10%, respectively; we also examined a scenario consisting of first and second year outplant mortality of 15% and 10%, respectively, plus 6% annual loss in aggregate outplant cover for years 3-20. In the baseline scenario, restocking increases fishery carrying capacity by 158.72% from 213.98 kg ha^{-1} in year 0 to 553.58 kg ha^{-1} in year 20. Depending on management regime, restocking increases optimal annual harvest between 45.50% (optimal fishery) and 82.99% (reserve) and fishery value between 13.05% (optimal fishery) and 67.79% (reserve). The presence of the marine reserve increases total harvest by 22.75% and fishery NPV by 50.90% from the optimal fishery.

Table 7. FRT reef fish biomass summary statistics

	Unprotected biomass (kg ha^{-1}) (n=595 transects)	Protected biomass (kg ha^{-1}) (n=202 transects)
Mean	230.63	335.79
Median	134.51	213.98
95th Percentile	730.75	1220.22
Std. Deviation	401.50	404.54

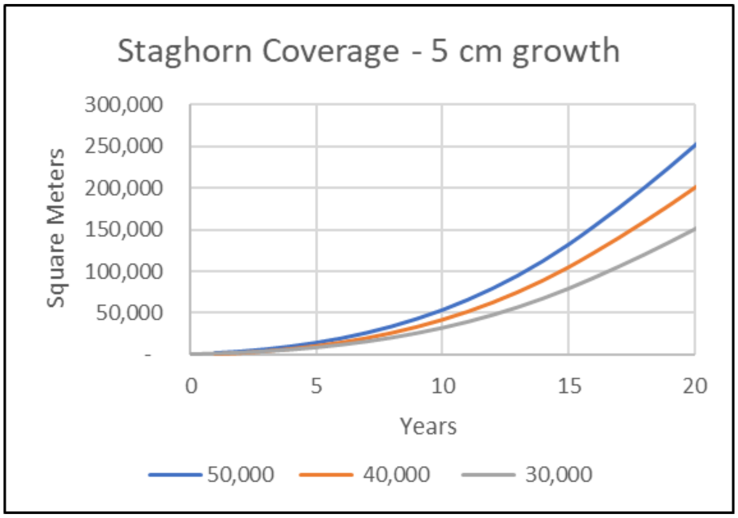

Figure 2. Staghorn coverage under alternative outplanting intensities

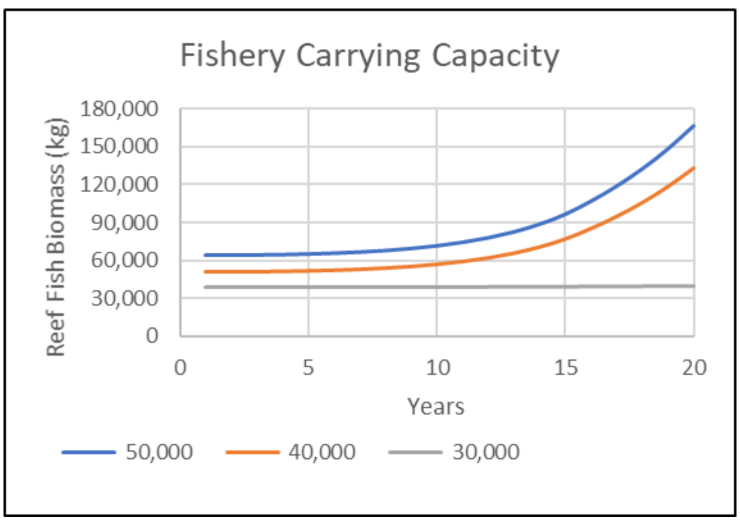

Figure 3. Fishery carrying capacity under alternative outplanting intensities

3.1 Optimally managed fishery

With no reserve, baseline fish harvest in year 20 is 17.01 kg ha^{-1} yr^{-1}, a 108.54% increase from $t = 0$ (8.16 kg ha^{-1} yr^{-1}) as a result of increased coral abundance and fishery productivity from restocking. Total fish harvest over 20 years is 71,215 kg (237.38 kg ha^{-1}), 45.5% greater than with no restocking (48,946 kg; 163.15 kg ha^{-1}). Fishery NPV is \$262.21 ha^{-1}, 13.05% greater than without restocking (\$231.94 ha^{-1}).

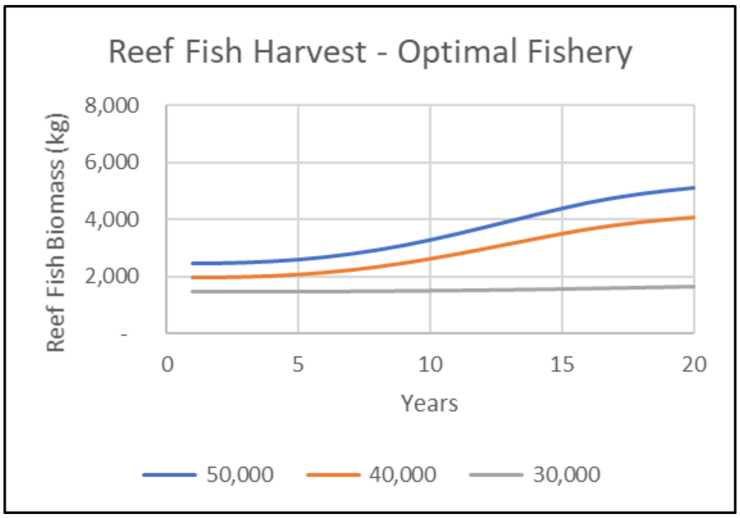

Figure 4. Fishery harvest under alternative outplanting intensities

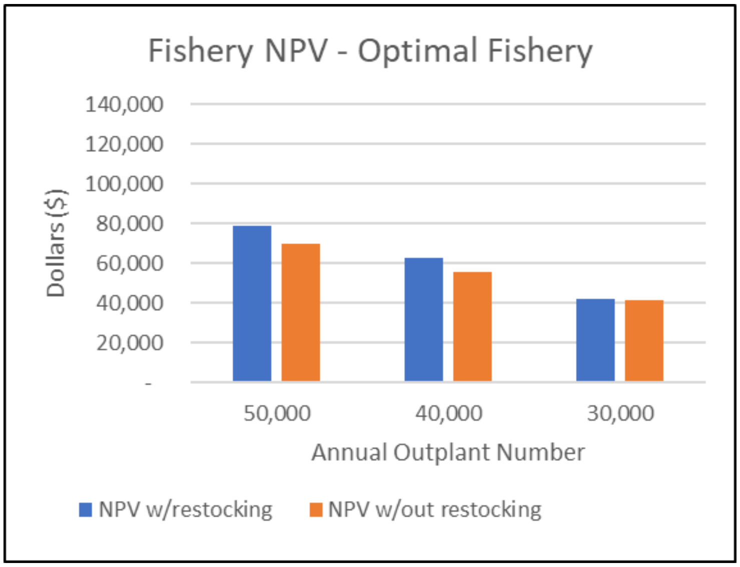

Figure 5. Fishery NPV – optimal fishery under alternative outplanting intensities

Table 8. Model results – 5 cm annual major axis growth

	Annual Outplant Volume				
	No Restocking	50,000 (Baseline)*	50,000 Increased mortality**	40,000	30,000
Year 20 carrying capacity (kg ha^{-1})	213.98	553.58	504.13	553.58	221.80
Optimal Fishery					
Harvest (kg ha^{-1})	163.15	237.38	233.62	237.39	170.03
NPV ($ ha^{-1})	231.94	262.21	260.59	262.21	234.68
Reserve – grounds					
Harvest (kg ha^{-1})	159.24	291.39	284.69	295.59	224.13
NPV ($ ha^{-1})	235.80	395.66	387.10	418.65	379.80

*Baseline mortality: 15% & 10%, respectively, in years one and two. **15% & 10% outplant mortality in years one and two, 6% annual loss of staghorn area in years 3-20. Per ha values are for a 300-ha fishery.

3.2 Marine reserve – grounds

With the marine reserve, baseline harvest in year 20 is 23.72 kg ha^{-1}, a 197.92% increase from $t = 0$ (7.96 kg ha^{-1}) and 39.44% greater than with no reserve; Total harvest over 20 years is 87,417 kg (291.39 kg ha^{-1}), 82.99% greater than with no restocking (47,773 kg; 159.24 kg ha^{-1}), and 22.75% greater than with no reserve (71,215 kg; 237.38 kg ha^{-1}). Fishery NPV is $395.66 ha^{-1}, an increase of 67.79% over $t = 0$ ($235.80) and 33.73% greater than without the reserve ($262.21 ha^{-1}).

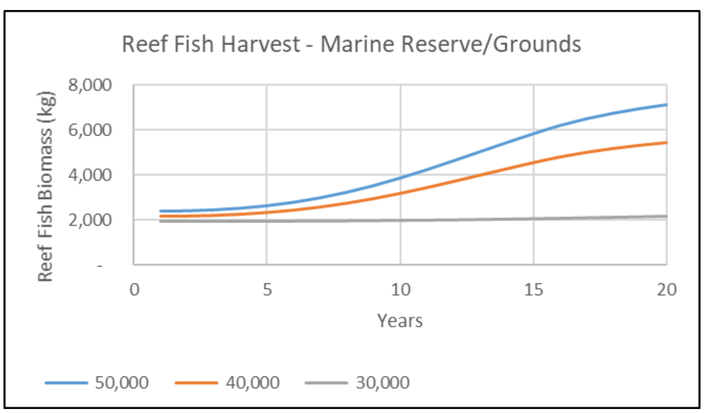

Figure 6. Fishery harvest under alternative outplanting intensities

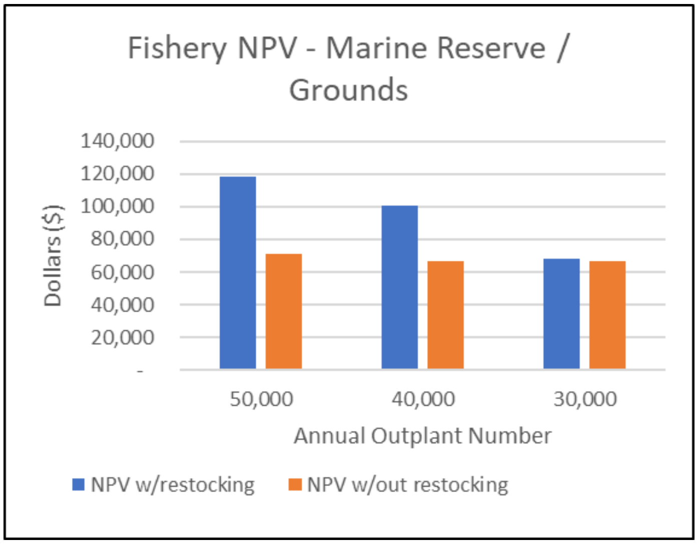

Figure 7. Fishery NPV with marine reserve under alternative outplanting intensities

4.0 Sensitivity analysis

4.1 Effect of outplant mortality rate

4.1.1 Carrying capacity

A decrease in the first- and second-year outplant mortality from baseline to 10% and 5%, respectively, (a 40% relative drop), increases the year 20 carrying capacity 18.25% from 553.58 kg ha^{-1} to 654.63 kg ha^{-1} (from 166,074 kg to 196,388 kg), relative to the baseline. An increase in first- and second-year mortality from the baseline to 20% and 15%, respectively, (a 40% relative increase) reduces year 20 carrying capacity by 21.66%, from 553.58 kg ha^{-1} to 455.01 kg ha^{-1} (166,074 kg to 136,504 kg).

Table 9. Reserve-grounds model results with various mortality rates

1st, 2nd yr. mortality	Fishery carrying capacity (kg)	Total harvest (kg)	Fishery NPV ($)
15%/10% (Base)	166,074	87,418	118,698
15%/10%/6%*	151,237	70,086	78,176
20%/15%	136,504	83,062	113,162
10%/5%	196,388	91,249	123,650

*First- and second-year mortality of 15% and 10%, respectively, and 6% annual loss in outplanted staghorn area in years 3-20.

4.1.2 Fishery harvest and profit: marine reserve - grounds

A decrease of 5% in the first- and second-year outplant mortality to 10% and 5%, respectively, results in an increased fish harvest over 20 years of 4.38% or 3,831 kg, over the baseline. The year 20 fish harvest is 2.75% greater (7,312 kg vs. 7,116 kg). Fishery profit increases 4.17% to \$412.17 ha^{-1} from \$395.66 ha^{-1}. A 5% increase in first- and second-year outplant mortality from the baseline to 20% and 15%, respectively, causes total fish harvest to decline 4.98%, or 4,356 kg over 20 years. Year 20 fish harvest is 6,803 kg vs. the baseline fish harvest of 7,116 kg, a difference of 4.40%. Fishery profit decreases 4.67% to \$377.21 ha^{-1} from \$395.66 ha^{-1}.

4.1.3 Fishery harvest and profit: optimally managed fishery

With no reserve, decreasing the first- and second-year outplant mortality to 10% and 5%, respectively, increases fish harvest over 20 years 3.02% from 237.38 kg ha^{-1} to 244.56 kg ha^{-1} (71,215 kg to 73,367 kg). Year 20 fish harvest with decreased outplant mortality is 5,213 vs. 5,103, an increase of 2.16%. With increased outplant mortality, total fish harvest is 229.23 kg ha^{-1} vs. 237.38 kg ha^{-1}, a decline of 3.43% (68,768 kg vs.

71,215 kg). Year 20 fish harvest is 4,926 vs. 5,213, a decline of 5.51%. With decreased mortality, fishery profit increases 1.19% to \$265.33 ha^{-1} from \$262.21 ha^{-1}. Increasing the outplant mortality reduces fishery profit by 1.33% to \$229.23 ha^{-1}. As with the reserve, marginal impacts of outplant mortality are greater when mortality decreases (vs. increases) from the baseline although less in the optimal fishery.

4.2 Effect of outplant volume

4.2.1 Carrying capacity

Decreasing outplant volume results in decreased total fishery carrying capacity, fish harvest, and profit, although per unit area results are mixed. Planting 40,000 colonies, fishery carrying capacity in year 20 is the same as the baseline at 553.58 kg ha^{-1}, while total carrying capacity falls to 132,859 kg from 166,074 kg. Planting 30,000 outplants, fishery carrying capacity falls 75.96% from the baseline to 221.80 kg ha^{-1} (39,924 kg).

4.2.2 Fishery harvest and profit: marine reserve – grounds

Decreasing outplant volume to 40,000 from the baseline (a 20% decrease), total harvest falls 18.85 % from 87,418 to 70,942, however, per-unit-area harvest increases 1.42% from 291.39 kg ha^{-1} to 295.59 kg ha^{-1}. With 30,000 colonies, total harvest falls 53.85% to 40,343; year 20 harvest declines 23.08% from the baseline to 224.13 kg ha^{-1}. When outplant volume is reduced from 50,000 to 40,000, total fishery NPV falls 15.35% from \$118,698 to \$100,477, however, per ha NPV increases 5.8% to \$418.65 ha^{-1} from \$395.66 ha^{-1}. Per ha NPV drops to \$379.80 ha^{-1} with 30,000 outplants, a decrease of

4.01% from the baseline; Total NPV falls 42.41% from the baseline to $68,364. The increase in unit-area-value and fish harvest from reducing outplant volume to 40,000 from the baseline is counterintuitive and, at least, partially due the higher marginal cost of harvest associated with the larger fishery. Costs increase 25% with the larger fishery (300 ha vs. 240 ha) while harvest increases 23%.

Table 10. Results with various growth rates and outplanting intensities

Marine Reserve - Grounds					
Annual growth (cm)	Annual outplants	Harvest (kg)	NPV ($)	Harvest before restocking (kg)	NPV before restocking ($)
5	50,000	87,418	118,698	47,773	70,739
5	40,000	70,942	100,477	43,620	66,459
5	30,000	40,343	68,364	39,120	66,712
4	50,000	78,972	108,132	47,773	70,739
4	40,000	65,121	92,982	43,620	66,459
4	30,000	40,024	67,929	39,120	66,712
3	50,000	69,628	96,778	47,773	70,739
3	40,000	58,680	84,929	43,620	66,459
3	30,000	39,725	67,523	39,120	66,712

4.2.3 Fishery harvest and profit: optimally managed fishery

Decreasing staghorn outplant volume to 40,000 from the baseline, unit area harvest remains the same as the baseline at 237.38 kg ha^{-1}. Fish harvest falls to 170.02 kg ha^{-1} planting 30,000 colonies annually, a decline of 57.02% from the baseline. Fishery NPV remains $262.21 ha^{-1} when outplant volume is reduced from 50,000 to 40,000, however, NPV drops to $234.68 ha^{-1} with 30,000 outplants, a decrease of 10.50%.

Table 11. Results with various growth rates and outplanting intensities

Optimal Fishery					
Annual growth (cm)	Annual outplants	Harvest (kg)	NPV ($)	Harvest before restocking (kg)	NPV before restocking ($)
5	50,000	71,215	78,663	48,946	69,583
5	40,000	56,973	62,931	39,156	61,666
5	30,000	30,605	42,243	29,369	41,751
4	50,000	66,471	76,662	48,946	69,583
4	40,000	53,177	61,331	39,156	61,666
4	30,000	30,280	42,113	29,369	41,751
3	50,000	61,222	74,512	48,946	69,583
3	40,000	48,978	59,611	39,156	61,666
3	30,000	29,975	41,992	29,369	41,751

5.0 Discussion

Using comparative statics, this study attempts to fill gaps in our understanding of how restocking and protecting staghorn populations on the FRT impacts the delivery and value of reef ecosystem services, namely support of commercial reef fish fisheries. We found that large-scale restocking and protection of staghorn populations may be effective in increasing commercially important reef fish carrying capacity, and optimal stocks, harvest, and fishery value. This result is consistent with literature examining the relationship between reef complexity and reef fish abundance and diversity (e.g., Clark and Edwards, 1998; Walker, *et al.*, 2009; Rogers *et al.*, 2014). In the optimally managed fishery, depending on treatment size, restocking increases fishery harvest and profit by as much as 45.50% and 13.05%, respectively. With the marine reserve protecting outplanted colonies, restocking increases fishery harvest and profit by as much as 85.63% and 77.50%, respectively, despite a 33.33% reduction in the size of the fishing grounds. The direction of these results, not necessarily the magnitude, are consistent with previous studies examining fisheries benefit of marine reserves (Roberts, *et al.*, 2001; Micheli, *et al.*, 2004; Jeffrey, *et al.*, 2012).

Because we introduce the fisheries benefit of the marine reserve only through the intrinsic growth rate of the stock (and not carrying capacity) in our model, management regime has no impact on fishery carrying capacity; if coral coverage is enhanced by protection, our model may underestimate the fisheries benefits. Cases of marine reserves supporting coral cover and resilience have been documented (i.e., Mumby and Harborne, 2010), however, results from studies are mixed. Examining three no-take reserves and three sites open to fishing in the Florida Keys, Toth, *et al.*, (2014) found that 14 years of protection did not influence coral cover. Huntington, Karnauskas, and Lirman (2011) found, after 10 years of protection, no clear indication of benefits to coral cover, colony size, or number of juvenile corals on Glovers Reef, Belize. Examining 10 sites in and outside of marine reserves in the Bahamas, Mumby and Harborne (2010) found rates of coral cover significantly higher in marine reserves than outside. Other studies have found declines in stony coral cover may continue for years after initial protection (Selig and Bruno, 2010). Many of the causes of staghorn declines on the FRT originate beyond the boundaries of marine reserves and jurisdiction of local resource managers. Water quality in the Florida Keys is impacted by inputs from the Everglades, Florida Bay, and the southwest Florida coast and rising ocean temperatures and acidification associated with global climate change are primary drivers of coral bleaching.

In the optimal fishery, per ha harvest and NPV from outplanting 40,000 colonies are the same for treatments of 50,000 colonies and dramatically higher than with 30,000. With the marine reserve, values decline slightly from 40,000 to 50,000 outplants. On the basis of per hectare harvest and profit values, treatments of 40,000 colonies (4 ha) are the most economically efficient of the three sizes examined. The marginal fishery benefits from protection are greatest with treatments of 30,000 colonies. At this level of outplanting, harvest and profit are 31% and 61% higher than without the marine reserve suggesting protection can be a vital component in the success of small restocking projects, in terms of fisheries benefits.

Our per hectare fishery values of \$234.68 to \$418.65, derived as the discounted value of the stream of revenues over the 20 year outplanting period, are consistent with previous studies finding annual US coral reef commercial fishery values ranging from \$36 to \$605 (2007 prices) (Brander and van Beukering, 2013). MacNeil *et al.* (2006) estimated, in the absence of fishing, global mean resident reef fish biomass should be 1,013 kg ha^{-1} (963, 1469); on heavily fished reefs, biomass was found to be 158 kg ha^{-1}. Derived from local abundance data, our optimal stock estimates range from 144 kg ha^{-1} to 153 kg ha^{-1} and baseline carrying capacities range from 572 kg ha^{-1} to 597.54 kg ha^{-1}. Our results appear low relative to McNeil and may reflect the relatively depleted state of the Keys commercial reef fish fishery.

Our model does not account for the contribution of outplant reproduction (sexual or asexual) to staghorn coverage. Staghorn coral has a propensity to reproduce asexually through fragmentation and colonies in the FKNMS have been observed spawning two years after outplanting. Reproductive output of staghorn corals is largely influenced by colony fecundity and population size and density (Knowlton, 2001) so marine reserves that support growth or survivorship of outplants may have substantial cumulative effects over the long term, which for model simplicity is not captured in our simulation, particularly considering the enhanced reproductive capacity of populations connected by restocking.

Colony mortality occurs only in years one and two in our model, 15% and 10%, respectively, in the baseline scenario (75% survivorship). Examining regional staghorn restoration programs, Schopmeyer, *et al.* (2017) found staghorn outplant survival to be 85.2+- 9.7% 12 months after transplanting (n=933 colonies); three programs that tracked mortality beyond the first year found two year outplant survivorship to be 75% (Schopmeyer, 2017). The marginal impact of mortality on carrying capacity, harvest, and NPV is greater when mortality increases from the baseline vs. when mortality decreases, suggesting marginal impacts to fisheries from changes in staghorn abundance are greater at lower levels of staghorn abundance.

We simulate outplant growth rates of three, four, and five cm $yr.^{-1}$. Typical staghorn growth rates range from 3.5 – 11 cm/yr. (Gladfelter, 1984) and more than 20 cm yr^{-1} has been observed (Tunnicliffe, 1983). A difference of 1 cm in annual colony growth rate may affect harvest over 20 years by over 10%, reinforcing the premise that the success and efficiency of restocking will be influenced by efforts to addresses local and global stressors affecting staghorn growth, health and resilience. With 30,000 outplants, harvest and NPV exhibit increasing returns as annual colony growth rate increases (i.e., the incremental increase in harvest and NPV grows as annual growth increases) whereas with 40,000 and 50,000 annual outplants, harvest and NPV exhibit diminishing returns to scale. This response is similar under both management regimes, although smaller with the optimal fishery, and again suggests impacts to fisheries from changes in staghorn coverage are greater at lower levels of staghorn abundance.

Limited knowledge of the linkages between staghorn coral and commercial reef fish stocks is a major obstacle to economic valuation of reef restoration efforts, particularly given the uncertainty surrounding the long term recovery path of outplanted staghorn colonies. Unlike in our simplified model, outplant growth and aggregate staghorn area resulting from restockng are likley to be non-linear and heterogenous across space and time as corals experience periods of growth and dieoff (Goergen *et al.*, 2019).

A comprehensive restocking program is underway to support remaining natural staghorn coral populations in SE Florida. Our results suggests large scale restocking and protection of staghorn corals can be effective in enhancing local fishery productivity and value. Although staghorn corals support other species on the FRT, we examine commercial reef fish because they are most commerically valuable and data existed to support our analysis. Therefore, the total benefit of outplant staghorn estimated in our study should be viewed as only a conservative, lower bound estimate.

Our study represents a first attempt to approximate the ecological and economic contribution to commercial reef fish fisheries from restocking staghorn coral populations on the FRT and contributes to the existing literature by establishing a general framework to examine the fishery impacts from restocking that may be applied to projects elsewhere. Results can inform decision making related to the management of Florida’s coastal resources, including the scale and intensity of restocking efforts and use of marine reserves to maximize returns. Quantifying the potential value of improved management can also support justification for scarce conservation funding. Ultimately, decisions related to large-scale restocking, particularly if coupled with marine reserves, will be made upon examination of many complex ecological and socio-economic issues likely to affect local, regional, and national stakeholders that rely on the coral reef ecosystems of the Florida Keys for their livelihood, recreation, and overall well-being.

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Chapter 2: Stated preference valuation of restocking and protecting the threatened staghorn coral on the Florida Reef Tract

1.0 Introduction

Staghorn coral (*Acropora cervicornis*) is a stony coral with antler-like cylindrical branches 0.25 to 5 cm in diameter that can form large thickets two to three meters tall and 30 meters long (NMFS, 2015). Staghorn is widely distributed throughout the western Atlantic, Caribbean, and Gulf of Mexico, including within the US jurisdictions of Puerto Rico and US Virgin Islands and four counties on the SE coast of Florida (Palm Beach, Broward, Miami-Dade, and Monroe) and, prior to the 1970s, was one of the most abundant structure building corals on shallow water Caribbean reefs for the past million years (Goreau, 1959; Geister, 1977; Adey, 1978; Jackson, 1992; Pandolfi and Jackson, 2001; Bruckner, 2002; Pandolfi, 2002;). Declines in staghorn abundance have been estimated as high as 97% regionally in the past four decades.

Staghorn coral's branching morphology provides essential habitat for fish and other organisms and a natural infrastructure protecting coastlines from damage associated with large tropical storms. Reef structural complexity has been linked to overall abundance and diversity of reef fishes (e.g., Grigg, 1994, Carpenter, *et al*., 1995; Lirman, 1999, Walker *et al*., 2009;), fish productivity, biomass, and reef carrying capacity (Warren-Rhodes *et al*., 2003; Graham *et al*., 2006). Fifty-percent of commercially important finfish species in Florida (e.g., amberjacks, groupers, hogfish, porgies, sea bass, snappers, tilefish, and triggerfish) use coral reef habitats during their lives, as do many recreationally targeted species (e.g., barracuda, dolphin, snook, tarpon, and trout) (Bruckner, 2002).

Currently, no other structure building coral species on the FRT provides the same type of complex habitat supporting these specific ecosystem functions, therefore, it is possible the continued loss of staghorn corals will result in significant loss in coral reef function and structure (Acropora Biological Review Team, 2005). The ecological and socio-economic consequences as Florida's staghorn populations have died off and reefs bio erode may be substantial (Done, 1996; Jones and Syms, 1998; Pittman *et al.*, 2007; Walker *et al.*, 2009).

1.1 Active restoration

Coral reefs have been declining globally over the previous five decades from local and global anthropogenic stresses, including overfishing, bleaching, and disease (Hughes *et al*., 2003; MEA, 2005). Wilkinson (2008) estimated 19% of coral reefs have been lost in the past three decades and another 35% are threatened with loss by 2050. Given the extent of the degradation, local conservation efforts and natural recovery may no longer be enough to preserve or restore the future health and integrity of the world's coral reefs (Goreau and Hilbertz, 2005). Practitioners and managers are increasingly relying on active coral reef restoration to counter patterns of decline and support the recovery of depleted coral populations and denuded reef ecosystems (Guzman, 1991; Rinkevich, 2005; Precht, 2006; Edwards, 2010; Johnson, *et al*., 2011; Schopmeyer *et al*., 2017). First practiced in the Indo-Pacific and Red Sea regions and now commonly used in Florida and the Caribbean, the "coral gardening" technique (Rinkevich, 1995; Johnson *et al*., 2011; Young *et al.*, 2012; Schopmeyer, *et al.*, 2017), entails removing live tissue from healthy coral colonies to be grown out in undersea nurseries (Rinkevich, 1995; Epstein *et al*., 2003; Shafir and Rinkevich, 2008; Shaish *et al*., 2008; Young *et al*., 2012). After approximately six months to one year in the undersea nursery, colonies are removed and “outplanted” close to one another on denuded reefs, so they spawn and help reseed surrounding reefs. Restocking is expected to increase sexual reproduction and support the long-term recovery of wild staghorn populations and their genetic diversity (NMFS, 2015) and each outplanting site directly enhances live coral cover, reef structural complexity, habitat, and economic value.

Rapid growth (3-11.5 cm yr^{-1}) (Gladfelter, 1983), high first survivorship (Schopmeyer, *et al*., 2017) and ability to reproduce asexually through fragmentation make staghorn well suited for restocking programs (Highsmith, 1982; Federal Register, 2008; Lirman, *et al*., 2010; NMFS, 2015). Young, *et al*., (2012) identified more than 60 *Acropora* restoration projects in 14 Caribbean and island nations and, currently, tens of thousands of nursery-reared staghorn colonies are being transplanted annually on reefs along the FRT off SE Florida as part of a comprehensive regional restocking program.

1.2 Marine reserves

Although some of the leading threats to corals currently approach being unmanageable (e.g., disease, rising ocean temperature, and hurricanes), minimizing those threats that are manageable may reduce overall stress and strengthen corals ability to recover from episodic stress events (NMFS, 2015). Research suggests no-take marine reserves protecting corals from damage associated with fishing, anchoring, and other physical stressors may enhance coral survivorship, recruitment and growth (Mumby *et al*., 2007; Selig and Bruno, 2010). Potential costs to extractive users from prohibiting fishing on restocked reefs include congestion on the remaining fishing grounds, increase in fuel costs, and user conflicts, for example, however, marine reserves may stabilize or increase stocks, inside their boundaries and maybe outside (McClanahan and Mangi, 2000) leading to reduced variability in total catch levels (Lauck *et al*., 1998) or enhanced long-run total catch (Sanchirico, *et al*., 2002). Benefits to non-extractive users may include increased biodiversity and the ability to enjoy a healthier marine ecosystem. Scuba divers and snorkelers have shown preference to coral reefs with high complexity, diversity, and abundance of fish and other marine organisms (Leeworthy and Wiley, 1997; Bhat, 2002; Leeworthy, *et al*., 2004).

Research has also shown that the health of coastal ecosystems is also important to individuals who may never intend to use the areas, but still value their existence (Peterson and Lubchenko, 1997; Brander and van Beukering, 2013). These non-use values often make up most of the total economic value of environmental goods like coral reefs.

Limited studies have attempted to measure the economic values that the public attributes to the restoration and protection of coral reef ecosystems, and none have focused their analysis on the threatened staghorn coral or recovery efforts in SE Florida. Such benefit estimates are required to undertake comparisons of the costs and benefit of alternative staghorn coral management strategies. Additionally, benefits estimates may provide insight into the level of public support for the restoration and protection of Florida’s coral reef ecosystems and the potential for alternative sources of financing for the restoration of Florida’s coastal resources.

1.3 Estimating the benefits of restocking staghorn populations

We used two attribute based stated preference methods to measure the total economic value of restocking and protecting populations of the threatened staghorn coral on the FRT. Stated preference methods are widely used in environmental valuation to collect information about respondent preferences for environmental amenities, typically through hypothetical scenarios presented in a survey format. The value a respondent places on a nonmarket environmental amenity can be derived from the maximum amount of money they would be willing to exchange for the delivery of that good or service, their maximum willingness-to-pay (WTP). Because SP techniques enable examination of preferences for levels of goods or services that differ from current levels or from levels that may have been observed previously, they are frequently the preferred approach for providing the economic valuation inputs required for cost-benefit analysis. Stated preference methods are also often the only approach to monetize the passive-use values of environmental amenities (Krutilla, 1967; Carson, *et al*., 1999). Because passive-use values contribute so much to the total economic value of some environmental goods, their examination is crucial for policymaking.

Contingent valuation (CV) and discrete choice experiments (DCE) are the most commonly used SP methods and generally contain choice sets, each comprised of a set of distinct hypothetical alternatives, from which respondents are requested to select their most preferred. For instance, alternatives are characterized by a set of attributes (one of which is generally cost), each taking one or more levels. The utility an individual derives from option j can be denoted

$$U_j(Q, I - A, X) = V_j(Q, I - A, X) + \varepsilon_j \qquad [18]$$

where Q denotes a vector of alternative specific attributes, I is the individual's disposable income, A is the amount the individual would be willing to pay for the improved environmental quality (e.g., coral abundance), and X is a vector of sociodemographic variables. The observable, or empirically measurable, component of utility is represented by $V_j(\cdot)$, while the unobservable stochastic component is denoted ε_j and modeled as econometric error. An individual is assumed to choose the alternative from which they derive the greatest utility (de Palma *et al*., 2008). That is, they would be willing to pay an amount A if,

$$V_1(Q_1, I - A, X) + \varepsilon_1 \geq V_0(Q_0, I, X) + \varepsilon_0 . \qquad [19]$$

By extrapolating WTP amounts to the population(s) benefitting from a change in environmental quality, the total economic benefit from that change can be approximated. To examine whether preference elicitation technique had an impact on respondent preferences and WTP estimates, our survey instrument contained two elicitation formats: a single-bound dichotomous choice CV format through which respondents were requested to select their preferred alternatives when presented with scenarios consisting of the status quo (SQ) and each of three alternative management interventions; and a DCE format, through which respondents were presented with the four alternatives and requested to rank them from most preferred to least preferred. To fit respondents' preferences into a utility-theoretic framework and estimate WTP, we use two specifications of logit models, conditional (CL) and rank ordered (RL).

Stated preference methods are commonly used to examine public preferences and WTP for enhancements in the status environmental services including air quality (Carson, *et al*., 1990), rivers and lakes (Carson and Mitchell, 1993; Cooper *et al*., 2004; Hanley *et* *al*., 2006), coastal waters (Georgiou *et al*., 1998; Hanley *et al*., 2003), coral reef habitats (Bhat, 2003; Bishop, *et al*., 2011), and marine biodiversity (Stefanski and Shimshack, 2015). Although commonly used in environmental valuation, SP methods are not without criticism. Respondent preferences and, thus, WTP values are contingent on the (generally limited) information possessed by the respondent and provided by the survey. Another perceived shortcoming is that because respondents typically possess limited knowledge on the functions of resources such as corals and coral reefs, value estimates do not reflect all ecological values.

1.4 The effect of risk perception on WTP

Previous studies suggest individual WTP for enhanced delivery of environmental goods or services is guided by socioeconomic variables like education, income, gender and familiarity or use of the resource being valued. Studies have shown that individual WTP for environmental improvements may also be dependent on perceptions and attitudes towards the risks associated with the issue in question. For example, Sukharomana and Supalla (1998) found WTP for enhancements to groundwater quality increased with the perception of the risks from exposure to contaminants. Georgiou *et al*. (1998) concluded WTP for improvements to coastal water quality has a strong positive correlation with the perceived health risks from exposure to contaminated waters, and Veronesi *et al*. (2014) found that climate change perception had a significant impact on individual WTP to mitigate flooding induced wastewater overflows into rivers and lakes. Studies have also shown there are commonly significant disparities between individuals' perceptions of risk and objectively quantified risk (Kraus *et al*., 1992; Campbell *et al*., 2002). Risks that are unfamiliar, uncontrollable, involuntary, irreversible, inequitably distributed, man-made, or catastrophic generally elicit the most concern (Slovic, 1987). Because perceptions of risk influence the decisions individuals make and frequently underlie disagreements over the optimal course of action, their consideration, and consideration of their underlying determinants, can help identify opportunities to inform people regarding actual risks and may reveal motives and barriers that stimulate or prevent action (Flynn, *et al*., 1994; Finucane, *et al*., 2000; Weber, Blais, & Betz, 2002).

We derive estimates of various psychometric risk measures that characterize people's risk perception (RP), risk concern (RC), and support of risk-reduction (RR) action, and examine whether, and to what extent, risk perception affects their WTP to support efforts to restock and protect Florida's staghorn corals. Following Hunter *et al*. (2012), our study incorporates psychometric measures into a conventional utility-theoretic model of non-market valuation and makes two notable contributions to the management of Florida's coastal resources. First, current research on the effects of risk perception is limited in coastal resource restoration and protection; results of our study provides insight on how different phases of risk evolution – RP, RC, and RR – influence the environmental value construct of individuals and WTP. Second, an understanding of the underlying determinants of risk perception can aid resource management agencies in efforts to engage the public and develop initiatives targeting awareness and literacy and, in turn, support for risk mitigation efforts like restocking and protecting staghorn corals (Vignola *et al*., 2013).

Research has shown geographic distance may also affect WTP for public goods with relatively large non-use values. Because distance impacts the use of environmental amenities (Sutherland and Walsh, 1985), empirical quantification of distance effects can be useful in decisions related to the aggregation of individual WTP values (Loomis, 1996) and decisions regarding sources of financing for environmental projects – for example, federal versus state or local funding (Concu, 2007). Multiple studies have discovered a negative relationship between distance and WTP values (Sutherland and Walsh, 1985; Loomis, 1996; Hanley *et al*., 2003) while others have found distance to be insignificant (e.g., Bateman and Langford, 1997; Pate and Loomis, 1997). Bateman and Langford (1997) found WTP to protect the Norfolk Broads, a destination for outdoor recreation in England, declined from its mean value as respondent distance increased from the Broads area. Pate and Loomis (1997) found distance influenced WTP for proposed programs to address environmental challenges in California and Sutherland and Walsh (1985) found distance and non-use values of water quality in Montana to be negatively correlated. Similarly, Georgiou *et al*. (1998) found a negative relationship between WTP to clean up a local river and geographic distance from respondents' residences to the project site. To examine whether geographic distance is a statistically significant determinant of respondent WTP for staghorn restocking and protection, we include the geographic distance from the centroid of the respondent's county of residence to Marathon, Florida in the Florida Keys as distance from as an explanatory variable in our valuation model. Finally, to enable examination into whether WTP estimates differ depending on the elicitation format and econometric analysis, we use two valuation methods: a conditional logit and rank-ordered logit.

2.0 Study background

The FRT stretches approximately 350 km southwest from Soldier Key in Biscayne Bay to the Tortugas Banks in the Gulf of Mexico. About two-thirds of the FRT lies within Biscayne National Park and the Florida Keys National Marine Sanctuary (FKNMS), a 9,900-square nautical km marine protected area (MPA) that surrounds the Florida Keys. Proximity to the Miami metropolitan area and Florida Keys has subjected the reef ecosystem to decades of intense human use. After years of declining water quality, episodes of coral bleaching and diseases, coral cover loss, and falling reef fish stocks, the FKNMS was designated in 1990 to protect the Florida Keys' coastal and marine resources. Leeworthy and Bowker (1997) estimated 13.7 million visitor days, worth annual non-market use value of over $1.2 billion, are spent annually in the Florida Keys, 75% of which is derived from natural resource-based activities like snorkeling, scuba diving and fishing. The inextricable linkages between the environment and economy make preservation and protection of existing resources critical to the future of the Florida Keys.

The dramatic loss of staghorn corals beginning in the 1970s has been largely attributed to white-band disease (Aronson and Precht, 2001), but linked to a multitude of inter-connected human induced and natural stressors. Today, most staghorn corals in the Florida Keys exist as isolated colonies or fragments on isolated patch reefs as opposed to their former abundance in deeper fore reef habitats (Miller *et al.*, 2008). Recruitment of new colonies has been observed at various locations in the Keys, but new recruits appear to be dying prior to reaching maturity. Bruckner (2002) found mean staghorn coverage on the FRT to be 0.049% with little variation among the eight habitat types surveyed.

Local fisheries have declined in productivity over the same period as staghorn corals; Twenty- three of 35 species of groupers, snappers, hogfish, and grunts have been chronically over-fished since the 1970s according to NMFS standards (Ault, 1998). In 1997, in response to user conflicts and resource degradation from concentrated visitor activity, 18 no-take sanctuary preservation areas (SPA), totaling 1.45 NM^2, were established in the FKNMS (NOAA, 2007). Additional, larger, protected areas were later established in the Dry Tortugas, a biologically rich area at the western end of the FKNMS. Almost 200 NM^2 in the Tortugas region is now closed to all fishing, however, in the time since designation of these protected areas, local staghorn populations have shown no evidence of natural recovery.

Having determined the threat of extinction was likely throughout all or much of staghorn's range, the NMFS listed staghorn coral as threatened under the ESA in 2006 (NMFS, 2006). The NMFS subsequently developed a recovery plan for the species with the goal of increasing the abundance and genetic variability of staghorn populations while reducing threats sufficiently to enable delisting. The plan details 24 recovery actions including improved fishing regulations (e.g., restricting fishing in areas near staghorn colonies) and active population enhancement through the implementation of a comprehensive restocking plan.

3.0 Methodology

Internet surveys have become common in research for the enhanced access and opportunities for data collection they offer along with the ability to quickly and conveniently administer complex instruments without personal interviewers or simultaneous interaction (Bishop *et al*., 2011). We administered a household survey in June 2017 through the Qualtrics online platform to elicit the preferences of residents of the Southeastern United States for restocking and protecting Florida's staghorn coral populations. Households in Florida, Alabama, Georgia, and Mississippi (n=3,135) were randomly selected from the Qualtrics panel to complete the survey. Of the 3,135 surveys initiated by respondents, 1,260 were completed in full. The survey contained a question to test whether respondents were reading the questions and providing thoughtful answers. Responses from respondents who "failed" this test question and completed the survey in less than the median respondent time were removed from consideration. We retained 1061 surveys for further analysis.

The questionnaire included four sections. The first section contained: (i) an explanation of the purpose of the survey, (ii) questions regarding respondents' familiarity and experience with coral reefs, and (iii) videos discussing ecosystem services provided by staghorn corals, status and threats facing staghorn corals and the FRT, and active efforts to recover lost staghorn populations. The second section included a brief recap of the status, threats, and efforts to restock staghorn populations discussed in the videos; The third section contained the choice model, and fourth section contained questions related to WTP motivations, risk perception, and socio-demographic characteristics. The 14 risk perception questions, following the psychometric paradigm developed by Slovic (Slovic, 1987), were rated on a five-point Likert scale. The first two sets of five questions evaluated respondents' perception of the anthropogenic risks facing Florida's coral reefs [Risk Perception (RP) variables] and respondents' concern [Risk Concern (RC) variables], respectively. The final four questions evaluated respondents' attitudes toward intervention or regulation [Risk Reduction or Regulation (RR) variables]. The data were tested for internal consistency of the questions in each group. Cronbach alpha values for RP, RC, RR groups and all questions combined were .93, .53, .66, and .87, respectively for the rank ordered logit dataset and .92, .50, .69, and .87 for the conditional logit dataset respectively, indicating an acceptable level of internal consistency.

In the choice model section, two techniques are used to elicit respondent preferences for four proposed staghorn coral management alternatives (SQ, and three alternative management interventions). One-half of respondents were randomly selected to rank the four management alternatives most preferred to least preferred and one-half were presented a dichotomous discrete choice format wherein the respondent was requested to choose sequentially between the SQ and each of the three alternatives with positive action. The purpose of using the two valuation methods was to allow examination into whether the WTP estimates differ depending on elicitation format and econometric analysis.

3.1 Valuation scenarios

In the survey instrument, each alternative was characterized in terms of its features or “attributes”. Described attributes include: (i) the number of staghorn colonies outplanted on the FRT annually and estimated coral cover resulting from the outplantings after 30 years, (ii) the area of new marine reserves protecting outplanted corals, and (iii) cost of each alternative to the respondent.

Attributes had two levels each: the SQ and a positive intervention. As summarized in Table 12, the outcomes were described in terms of staghorn area on the

FRT after 30 years. The specific spatial and biological parameters that characterized the alternatives were simulated using the staghorn coral growth model described in Chapter one. To account for substitution and income effects (Arrow *et al.*, 1993), the survey contained verbiage advising respondents to keep in mind that paying for the intervention would leave less funds for other things that the respondent's household may have needed. The proposed payment vehicle was an additional annual tax added to the annual federal income tax obligation. The sample included two sub-sets of respondents, those who had visited a coral reef in Florida in the past three years and those who had not, allowing us to determine whether the non-use component of the total coral economic valuation was significant. Questions also examined whether respondents understood the alternative programs and confidence in their potential effectiveness.

The choice model section of the survey contained a SQ alternative which assumed the current level of outplanting (approximately 50,000 colonies yr^{-1}) would continue for at least 30 years with no new marine reserves to protect outplanted colonies. In addition to the SQ, there were three alternative programs in the survey: (1) increase staghorn outplants on the FRT from the current annual average of approximately 50,000 to 300,000, (2) implement no-take marine reserves to protect the 50,000 colonies currently outplanted every year, (3) increase staghorn outplants on the FRT from the current annual average of approximately 50,000 to 300,000 and implement no-take marine reserves to protect outplanted corals.

3.2 Respondents' risk perception

Because of multi-collinearity, the responses to the 14 RP questions could not be used as explanatory variables in the WTP model. To identify the factors accounting for the most variation in the observed responses and enable their inclusion in the WTP model, we conducted exploratory factor analysis (EFA) on the 14 RP variables. The varimax (orthogonal) rotation was used to extract the factors (DiStefano, 2009). Results suggested two meaningful factors with eigenvalues >1; variables were assumed to load on a factor if the loading exceeded 0.5. Using these criteria, we associated the following statements, or attitudes, with the factors they loaded on (1) "willingness to reduce risk" (WRR) and (2) "unwillingness to worry about risks" (UWR).

Regression factor scores were predicted for the two factors with eigenvalues >1 using a least squares regression approach (Thurstone, 1935) in which the regression equation independent variables are the standardized observed values of the items (i.e., respondent responses) in the estimated factors. These independent variables are weighted by regression coefficients calculated as the product of the inverse of the observed variable correlation matrix and matrix of factor loadings (DiStefano, 2009). Regression factor scores predict the location of each respondent on the factor and have been shown to be unbiased when used as independent variables in regression models (Devlieger, Mayer, and Rosseel, 2016). A similar approach was used to examine the public's WTP for enhancements to lake water quality (Cooper *et al*., 2004), reductions in risks from exposure to cyanobacterial blooms (Hunter *et al*., 2012), and recovery of endangered species (Aldrich *et al*., 2007).

3.3 Econometric models

We apply the standard conditional logit model (McFadden, 1973) to the dichotomous choice dataset and rank ordered logit, a generalization of the CL, to the dataset of ranked alternatives (Hausman and Ruud, 1987). Conditional logit models allow choices among alternatives to be modeled as a function of the attributes of the alternatives in the choice set as well as the characteristics of the individual making the choice. In the standard CL model, individuals are assumed to select the alternative from the choice set that provides the greatest utility. Following the random utility model in equation (2), in the standard CL, the probability of a respondent saying "yes" to paying amount A is

$$Prob(Yes\ to\ A) = Prob[V_1(Q_1, Y - A, X) + \varepsilon_1 \geq V_0(Q_0, Y, X) + \varepsilon_0] \quad [21]$$

$$= Prob[V_1(Q_1, Y - A, X) - V_0(Q_0, Y, X) \geq \varepsilon_0 - \varepsilon_1] \quad [22]$$

$$= Fn(n) \quad [23]$$

$$= Fn(\Delta V) \quad [24]$$

where $n = \varepsilon_0 - \varepsilon_1$ and $\Delta V = V_1(Q_1, Y - A, X) - V_0(Q_0, Y, X)$, the difference in utility between the two alternatives, and $F_n(\Delta V)$ is the cumulative probability density function. Per the logit model

$$F_n(\Delta V) = \frac{1}{1+e^{-\Delta V}} \quad [25]$$

$$F_n(\Delta V(A)) = \frac{1}{1+e^{-\Delta V(A)}} \quad [26]$$

The observable component of utility V_k for each respondent is specified to be linear in parameters, such that

$$U_{ri} = \sum_k \beta_{rik} X_{rik} + \varepsilon_{ri} \quad [27]$$

where X_{rk} is a vector of K choice-related characteristics consisting of individual characteristics and observed attributes, and B_{rk} is a vector of K coefficients to be estimated. In the RL, the probability individual i will select program k in round one of the ranking process can be denoted

$$Prob(individual\ i\ chooses\ program\ k) = P_{ik} = P\left(U_{ik} > U_{ij}, for\ all\ j \neq k\right)$$

$$= P\left(V_{ik} + \varepsilon_{ik} > V_{ij} + \varepsilon_{ij}, \forall j \neq k\right) \qquad [28]$$

$$= P\left(\varepsilon_{ij} - \varepsilon_{ik} < V_{ik} - V_{ij}, \forall j \neq k\right) \qquad [29]$$

In this study, respondents make a choice among four alternatives: the SQ and three with some increase in the abundance of staghorn corals compared to the SQ. This increased abundance of staghorn coral can be realized at a cost to be paid as an addition to the respondents' annual federal income tax obligation, and the cost of maintaining current abundances of staghorn corals is zero. From this, equation 18 can be generally formulated as

$$U_{ij} = (\beta_{MR}MR_j + \beta_C C_j + \beta_{MRC}MR_j C_j)X_i + \varepsilon_{ij} \qquad [30]$$

where i denotes individual respondents (i = 1…n); j denotes the four program alternatives in the survey (1 = SQ, 2 = Marine Reserve Program, 3 = Staghorn Restocking Program, and 4 = the combination of programs 2 and 3); X_i is a k × 1 vector of individual specific variables, including a "1" to enable consideration of alternative-specific constant (ASC) terms; MR_j and C_j are scalar variables indicating whether or not marine reserves or staghorn restocking programs appear in alternative j; and β_{MR}, β_C, and β_{MRC} are 1 x k vectors of the marginal contributions to individual utility from the respective programs.

Seven specifications of this model were estimated to explore the effects of individuals' socio demographic characteristics and risk perceptions on WTP in a stepwise fashion. Model 1 is specified with the full set of individual specific covariates interacted with the ASC.

$$U_{ij} = \beta_{MR}MR_j + \beta_C C_j + \beta_{MRC}MR_j C_j + \beta_{Aedu}A_{ij}edu_i + \beta_{Ainc}A_{ij}inc_i + \beta_{Atimes}A_{ij}times_i + \beta_{Aage}A_{ij}age + \beta_{Agender}A_{ij}gender + \beta_{AFLres}A_{ij}FLres + \beta_{AWRR}A_{ij}WRR + \beta_{AUWR}A_{ij}UWR + \beta_{Aenviro}A_{ij}enviro + \beta_{Adist}A_{ij}dist + \varepsilon_{ij} \quad [31]$$

where A is ASC, β_{Aedu} is the vector of coefficients from the interaction of ASC and Edu, β_{Ainc} is the vector of coefficients to the interaction of ASC and Income, β_{Atimes} is the vector of coefficients to the interaction of ASC and times, β_{Aage} is the vector of coefficients to the interaction of ASC and Age, $\beta_{Agender}$ is the vector of coefficients to the interaction of ASC and gender, β_{FLres} is the vector of coefficients to the interaction of ASC and flres, β_{AWRR} is the vector of coefficients to the interaction of ASC and WRR, β_{AUWR} is the vector of coefficients to the interaction of ASC and UWR, $\beta_{Aenviro}$ is the vector of coefficients to the interaction of ASC and enviro, and β_{Adist} is the vector of coefficients to the interaction of ASC and distance.

Model 2 is the original choice model with the distance variable removed, Model 3 is Model 2 with the variable reflecting the number of times the respondent visited a coral reef in the previous three years removed, Model 4 is Model 3 with the age variable removed, Model 5 is Model 4 with the variable resident variable removed, Model 6 is Model 5 with education variable removed, and Model 7 is Model 6 with the WRR and UWR variables removed. The model variables for the CL and RL models are defined and described in Tables 13 and 19, respectively and estimation results presented in Table 16 and Table 26, respectively. In addition to the final model presented here, we explored several model specifications and found that some led to results that differed significantly from our final model. Specifically, early runs of the CL model included the variables for the number of times a respondent has visited a coral reef, age, gender, distance, question sequence and whether the respondent was a Florida resident, but these were found to be insignificant and removed from the final model to improve estimation efficiency; A similar procedure was followed with the RL models. Estimation of mean WTP is not significantly impacted by the inclusion or exclusion of the omitted covariates. The conditional logit and rank order logit model variables are summarized in Tables 16 and 25, respectively.

4.0 Results

4.1 Descriptive statistics

Examination of respondent demographic information for the two sets of respondents (Tables 14 and 20), confirms, other than gender, the samples are generally representative of the national and respective state populations. The mean age of US and Florida residents is 38 and 41years old, respectively, compared to mean of 35 and 36 and median of 31 and 33 years for the CL and RL datasets, respectively. The distribution of race across respondents is representative of the SE United States. In Florida, where approximately one-half of the survey respondents resided at the time the survey was administered, the population is 16.9% black, 54.1% white, and 25.6% Hispanic. Compared only to the Florida population, whites appear to be overrepresented and Hispanics underrepresented among respondents, however, in AL, MS, an GA, where one-half of respondents resided, whites and Hispanics make up a smaller percentage and blacks a larger percentage of the overall population, likely explaining much of the difference. Mean respondent per capita income is $25,414 and $26,803 for the RL and

CL datasets, respectively, compared to $29,829 nationally and between $21,651 - $27,598 for the states of FL, GA, MS, and AL. Nationally, 87.0% and 30.3% of individuals over the age of 25 graduated from high school or higher and college or higher, respectively. Of the CL dataset respondents over the age of 25, 68.44% were graduates of high school or higher, and 44.27% were graduates of college or higher; 64.89% of the RL respondents were graduates of high school or higher and 47.84% graduates of college or higher. In both samples, the proportion of female respondents is dramatically higher than in the US population.

4.2 Rank ordered logit risk perception data and factor analysis results

Results of the psychometric questions are presented in Table 22. Respondents indicated they are not strongly convinced that Florida's coral reefs have deteriorated dramatically in recent decades or that the risks to Florida's coral reefs and fisheries will continue to increase into the future, with mean scores of 3.93 and 3.81, respectively. Respondents indicated they were relatively comfortable with the level of risks facing Florida's coral reefs and marine resources, with a mean score of 3.71, but indicated they are uncertain whether future generations will address the risks faced by Florida's reefs appropriately (mean score of 2.67) or whether the health of Florida's coral reefs is managed by the relevant authorities (mean score of 2.97). On average, respondents indicated they were between "moderately" and "very" concerned about the health and future of Florida's coral reefs and coastal resources with mean scores ranging from 3.53-3.78. Overall, results suggest moderately high level of support for regulatory action (mean score 4.05) as well as a moderately high sense of individual responsibility for contributing toward the protection and enhancement of coral reefs (mean score 3.90).

Examination of risk responses across different demographic groups suggests that the respondent's gender and whether they were a Florida resident had no significant impact on their RP, RC, or RR . Overall, level of education is positively correlated with RP, RC, and RR. Notably, income is a statistically significant determinant of RR, but not of RP and RC while the number of times a respondent has visited a coral reef is a determinant of RP and RC, but not RR. Race was a statistically significant determinant of RP and RR, but not RC. Responses to the psychometric questions were examined further using exploratory factor analysis (EFA). Kaiser-Meyer-Olkin values indicated that all 14 variables were suitable for inclusion (all values >0.60, overall value 0.9123). Two meaningful factors (eigenvalues >1) were extracted through a varimax (orthogonal) rotation, suggesting respondents' RP, RC, and RR were determined by two underlying, or latent, factors. The groups of variables contained in the two factor groupings were labeled "willingness to reduce risk" (WRR) and "unwillingness to worry about risk" (UWR) for factors one and two, respectively. Observed risk variables used in the EFA and their corresponding loadings are represented in Table 24. All five of the RP questions are contained in factor one and had large, positive loadings (>0.7959) on that factor, indicating it describes the variation in those variables adequately. Two of the five RC, and three of the four RR, questions are contained in factor one. The RC questions in this factor elicit the level of concern for the general health of Florida's coral reefs and for coral bleaching associated with climate change; the three RR questions express support for the protection and enhancement of Florida's coral reefs.

Factor two contains questions that address specific threats, (i.e., overfishing, marine pollution, biodiversity, and physical damage to coral reefs and sea grass beds) that are commonly understood by the public and generally considered to be manageable. Also contained in factor two was the statement: “The relevant public agencies will manage Florida’s coral reefs without my contribution to the effort”, suggesting that whether and to what extent a threat is perceived to be locally manageable may be correlated with respondents’ confidence in the ability of public agencies to manage them and, therefore, a reduced RR.

To examine correlation between the three risk categories, Pearson’s correlation coefficients were estimated for their sums of scores. The correlation coefficient between RP and RC of 0.3569 ($p < 0.001$) exhibits a moderately strong and statistically significant positive correlation between RP and RC. As would be expected, the correlation coefficient between RC and RR is strong (0.6741, $p < 0.001$) and positive. This supports the hypothesis that respondents who indicate a high level of concern for the risks facing Florida’s coral reefs are more likely to support and express WTP to protect coastal resources and mitigate risk. Interestingly, the correlation between RP and RR (0.5104, $p < 0.001$) is stronger than the correlation between RP and RC, suggesting a direct pathway from RP and RR for some respondents.

4.3 Respondents’ WTP and effect of risk perception

The results of the CL model are presented in Table 16. WTP was estimated

$$WTP_{ij} = \frac{-(\hat{\beta})X_i}{\hat{\beta}_{cost,j}} \quad [32]$$

where $\hat{\beta}$ is a vector of coefficients for the individual specific covariates, and $\hat{\beta}_{cost,j}$ is the estimated cost coefficient for program j. The sign of the cost coefficient is negative for all three alternatives as expected but significant only for the marine reserve program, implying a measurable propensity to choose only the marine reserve program (and not the restocking program or the alternative combining the restocking and marine reserve program) over the SQ apart from any propensity explained by the other model covariates. Because only the cost coefficient for the marine reserve program was significant, WTP was estimated only for the coral and combined alternatives. The coefficient for distance was not significant for any of the three alternatives, however, we estimated WTP with and without distance as a covariate for comparison. Household WTP estimates are presented in Tables 17 and 18. Both risk-related factor variables were positive and statistically significant, indicating respondents' attitudes toward and perceptions of the risks facing Florida's coral reefs had a positive and significant impact on the probability of choosing all three of the programs to restore and protect staghorn coral populations. The coefficient for income was positive for all three programs, but significant only for the coral restocking program, implying income has a positive and significant impact on the probability of a respondent selecting the coral restocking program but that no significant income effects exist for the combined and marine reserve programs. The coefficient for enviro is positive for all three alternatives but significant only for the combined program, implying that whether someone self identifies as a "strong" or "very strong" environmentalist affects the probability of whether they select the combined program but not the coral restocking or marine reserve programs, individually. The coefficient for education is significant and positive for the marine reserve and both programs, implying it is not a significant determinant of whether the respondent selected the coral restocking program. The coefficient for the variable indicating question sequence was not significant, suggesting the order in which the alternatives were presented to respondent was not a significant determinant of respondent preferences.

Table 12. Alternative programs and outcomes

Management alternative	Annual outplants	Marine reserves to Protect outplants?	Staghorn area after 30 yrs. (sq. miles)
Status quo	50,000	No	.5
Restocking	300,000	No	4
Marine reserves	50,000	Yes	1
Combined	300,000	Yes	5.5

Table 13. Definition of variables included in the conditional logit model.

Alternative-specific variables	**Variable definition**
Coral	A variable indicating the restocking program appeared in the chosen alternative
Marine reserve	A variable indicating the marine reserve program appeared in the chosen alternative
Cost	The cost to the household of the alternative
Individual-specific variables	**Variable definition**
WRR	Risk Factor Score 1
UWR	Risk Factor Score 2
Enviro	A dummy variable that equals 1 if the respondent indicated they were either a "very strong" or "strong" environmentalist
Edu	A variable indicating the level of respondent education. 1=Less than high school, 2=HS grad, 3= Some College, 4=College Grad.
Inc	Respondent household per capita income
Age	Respondent age
Dist	Distance from location survey was completed to the Florida Keys Marathon International Airport, located approximately in the middle of the Keys island chain.
Gender	Respondent gender

Table 14. Conditional logit respondent demographics

	n	Mean	Median	Std. Dev.	Min	Max
Age	529	34.59	31	13.65	16	79
Household size	529	2.75	3	1.27	1	5
Per capita income ($000's)	529	26.80	17.50	28.07	10	250
Gender	*n*	%				
Female	365	69				
Male	164	31				
Race/Ethnicity	*n*	%				
White	312	58.98				
Hispanic	75	14.18				
Black or African-American	121	22.87				
Other	21	3.97				
Education	*n*	%				
Less than high school	20	3.78				
High school graduate	143	27.03				
Some college	157	29.68				
College graduate	209	39.51				

Table 15. Summary of variables included in the final conditional logit model (n=529)

Variable	Mean	Standard	Min.	Max.
Coral cost ($/household)	118.0	53.829	50	200
MR cost ($/household)	106.7	43.732	40	160
Both cost ($/household)	213.3	64.832	85	340
WRR	0.000	0.9693	-3.8920	4.4382
UWR	0.000	0.8258	-2.8092	2.5058
Enviro	0.279	0.4493	0	1
Education	3.049	0.9031	1	4
Income ($K/person)	26.80	28.069	2	250

Table 16. Results from conditional logit

	Both	Marine Reserve	Coral
Cost	-0.0016	-0.0053***	-0.0022
	(0.0011)	(0.0018)	(0.0014)
Enviro	0.6336***	0.1452	0.2942
	(0.2448)	(0.2310)	(0.2220)
WRR	0.7551***	0.7069***	0.5730***
	(0.1138)	(0.1127)	(0.1079)
UWR	0.3992***	0.4327***	0.3289***
	(0.1254)	(0.1186)	(0.1150)
Edu	0.1402*	0.1414*	-0.0044
	(0.0847)	(0.0737)	(0.0685)
Income	0.0045	0.0037	0.0071*
	(0.0038)	(.00367)	(0.0037)
Observations	529	529	529
Wald chi^2	79.68	64.13	50.17
Prob > Chi^2	0.0000	0.0000	0.0000
Log-likelihood	-313.3607	-327.7004	-337.6880

Standard error in parentheses; *p<0.10, **p<0.05, ***p<0.01

Table 17. Household WTP without distance as a covariate

Model	WTP	Std. Err.	z	Prob > |z|	95% Conf. Interval	
Both	457.24*	187.19	2.44	0.015	90.35	824.13
Marine reserve	107.89	17.29	6.24	0.000	74.01	141.78
Coral	115.33*	40.02	2.88	0.004	36.88	193.78

*Logit model cost coefficient not statistically significant

Table 18. Household WTP with distance as a covariate

Model	WTP	Std. Err.	z	Prob > |z|	95% Conf. Interval	
Both	441.09*	168.09	2.62	0.009	111.64	770.54
Marine Reserve	105.89	18.27	5.79	0.000	70.07	141.71
Coral	112.62*	41.25	2.73	0.006	31.77	193.47

*Logit model cost coefficient not statistically significant

4.4 Rank ordered logit

We estimated seven rank ordered logit models in which individual-specific variables were interacted with the ASC terms to generate variation across alternatives necessary for estimation. Summary statistics for model variables are presented in Table 21. Results of the RL model are presented in Table 26 and discussed below. A Wald test on the eight final model covariates cannot reject their joint significance ($X^2_{(21)} = 220.12$, $p<0.001$). The pseudo simulated log-likelihood at model convergence is:-1564.776.

Model one contains all socio demographic variables generated through the survey instrument interacted with the indicator terms. In subsequent models, we removed the interaction variables containing dist, times, age, flres, gender, and educate one at a time, re-estimating the model with each removal. As expected from economic theory, the coefficient for bid is negative and significant in all seven models. Household WTP was estimated

$$WTP_{ij} = \frac{-(\beta_C C_j + \beta_{MR} MR_j) X_i}{\beta_{cost,j}} \quad [33]$$

where i represents the individual survey respondents ($i = 1 \ldots n$); j represents the four program options in the survey (1 = SQ, 2 = the marine reserve program, 3 = the staghorn restocking program, and 4 = the combination of programs 2 and 3), C_j and MR_j are scalar variables indicating whether stocking or marine reserves are in alternative j, X_i is a vector of individual specific variables, and $\beta_{cost,j}$ is the coefficient for the cost of program j.

Household WTP was estimated using all seven models to examine the impact of individual covariates on mean preferences; WTP estimates for the restocking program, marine reserve and combined programs ranged from \$94.74 to \$179.01, \$.03 to \$96.60, and \$96.00 to \$275.61, respectively, and reflect substantial variation across models. The insignificance of the variable representing the number of times a respondent had visited a coral reef implies non-users maintain a significant WTP for coral restoration and protection. The coefficient for the ASC term for the coral program is positive and insignificant in every model other than in model six, where it is positive and significant, and the coefficient for the indicator variable for marine reserve is negative and insignificant in every model, other than in model six where it is positive and insignificant. These results imply that other than in model six, there is no measurable propensity to select an alternative including restocking or marine reserves over the SQ beyond any propensity explained by the other model covariates. The coefficients for the variables of income and enviro interacted with coral are positive and significant implying that respondent income and whether they identify themselves as a “strong environmentalist” or “very strong environmentalist” has a significant and positive impact on the probability they select a program with coral in it. The coefficient for the variable interacting income with the marine reserve program is not significant, suggesting no significant income effects exist for either of the alternatives with marine reserves. This may because cost of the marine reserve program was generally the least-cost alternative and presented a smaller financial burden on households. The coefficient for the variable interacting gender with coral is negative and significant implying that the presence of coral in the alternative reduced the probability that females would select that alternative. The WRR and UWR variables interacted with coral and marine reserve are positive and significant ($p < .001$) implying that respondent risk characteristics are positively correlated to WTP for both interventions.

Finally, a weighted risk-adjusted WTP was estimated (Table 28) using the sample average percent of respondents that expressed different levels of agreement to risk attitudes as weights. On average, 5.73% strongly disagreed (Likert scale =1), 10.78% somewhat agreed (2), 28.54% neutral (3), 27.90% agreed (4), and 27.05% strongly agreed (5) to the 14 risk questions. Risk adjusted WTP for coral is approximately 15% less ($155) than unadjusted WTP, and risk adjusted WTP for the marine reserve and both alternatives are 129% ($22.05) and 37% higher ($377), respectively.

Table 19. Definition of variables included in the rank ordered logit model

Alternative-specific variables	**Variable definition**
Coral	A variable indicating the restocking program appeared in the chosen alternative
Marine Reserve	A variable indicating the marine reserve program appeared in the chosen alternative
Cost	The cost to the household of the alternative
Individual-specific variables interacted with the restocking program	
Times x coral	Number of times respondent has visited a coral reef interacted with alternatives that include the restocking program, 0 otherwise
Age x coral	Respondent age interacted with alternatives that include the restocking program, 0 otherwise
Gender x coral	Gender interacted with alternatives that include the restocking program, 0 otherwise
Flres x coral	A dummy variable that equals 1 if the respondents is a resident of Florida, interacted with alternatives that include the restocking program, 0 otherwise
WRR x coral	Risk Factor Score 1 interacted with alternatives that include the restocking program, 0 otherwise
UWR x coral	Risk Factor Score 2 interacted with alternatives that include the restocking program, 0 otherwise
Enviro x coral	A dummy variable that equals 1 if the respondent indicated they were either a “very strong” or “strong” environmentalist interacted with alternatives that include the restocking program.
Dist x coral	Geographic distance from the location where the survey was completed to the middle of the Florida Keys interacted with alternatives that include the restocking program.
Individual specific variables interacted with the marine reserve program	
Times x marine reserve	Whether a respondent has visited a coral reef interacted with alternatives that include the marine reserve, 0 otherwise
Age x marine reserve	Respondent age interacted with alternatives that include the marine reserve program, 0 otherwise
Gender x marine reserve	Gender interacted with alternatives that include the marine reserve program, 0 otherwise
Flres x marine reserve	A dummy variable that equals 1 if the respondents is a resident of Florida, interacted with alternatives that include the marine reserve program, 0
WRR x marine reserve	Risk Factor Score 1 interacted with alternatives that include the marine reserve program, 0
UWR x marine reserve	Risk Factor Score 2 interacted with alternatives that include the marine reserve program, 0
Enviro x marine reserve	A dummy variable that equals 1 if the respondent indicated they were either a “very strong” or
Dist x marine reserve	Geographic distance from the location where the survey was completed to the middle of the Florida Keys interacted with alternatives that include the marine reserve program.

Table 20. Rank ordered logit respondent demographics

	n	Mean	Med	Std. Dev.	Min	Max
Age	530	35.79	33	13.72	16	85
Household size	530	2.91	3	1.288	1	5
Per capita income ($000's)	530	25.41	17.5	24.76	10.0	250.0
Gender	*n*	%				
Female	398	75.10				
Male	132	24.90				
Race/Ethnicity	*n*	%				
White	362	68.3				
Hispanic	52	9.81				
Black or African-American	95	17.92				
Other	21	3.96				
Education						
Less than high school	16	3.02				
High school graduate	126	23.77				
Some college	180	33.96				
College graduate	208	39.25				

Table 21. Summary of variables included in rank ordered logit model (n=530)

Variable	Mean	Std. Dev.	Min.	Max.
Income	25.414	24.761	2	250
Times	1.6111	3.4709	0	30
Age	35.797	13.730	16	85
Gender	1.7509	0.4326	1	2
FLres	0.6528	0.4762	0	1
WRR	0.0000	0.9703	-3.1130	1.4328
UWR	0.0000	0.8036	-2.2427	2.2690
Enviro	0.26037	0.4389	0	1

Table 22. Risk perception, risk concern, and attitudes toward risk reduction

Perception of Risks	***n***	**Mean**	**Std. Dev.**
(1) Florida's coral reefs have deteriorated dramatically in recent decades.	530	3.93	1.08
(2) I am comfortable with the level of risks facing Florida's coral reefs and marine resources.	530	3.71	1.17
(3) The health of Florida's coral reefs is managed by the relevant authorities.	530	2.97	.94
(4) The risks to Florida's coral reefs and fisheries will continue to increase into the future.	530	3.81	1.03
(5) Future generations will address the risks faced by Florida's reefs appropriately	530	2.67	1.08
Concern about specific risks	**n**	**Mean**	**Std. Dev.**
(6) Regarding the health of Florida's coral reefs	530	3.55	1.09
(7) Regarding overfishing in Florida and other US states/jurisdictions	530	3.53	1.12
(8) Regarding marine pollution and loss of marine biodiversity	530	3.78	1.10
(9) Regarding rising ocean temperatures and bleaching of Florida's corals	530	3.69	1.14
(10) Regarding physical damage to coral reefs and sea grass beds.	530	3.72	1.09
Risk reduction or regulation	***n***	**Mean**	**Std. Dev.**
(11) Government agencies must start to take actions to preserve and protect Florida's coral reef ecosystems.	530	4.05	1.08
(12) As a citizen, I am also responsible for contributing towards the protection and the enhancement of coral reefs.	530	3.90	1.02
(13) Any human activities that adversely affect the health of coral reefs and fish populations should be regulated.	530	3.99	1.07
(14) The relevant public agencies will manage Florida's coral reefs without my contribution to the effort.	530	3.08	1.20

Table 23. Risk perception, concern, and reduction preferences across demographic groups

Demographic Characteristics	Levels	*n*	RP (Out of a max score of 25)	RC (Out of a max score of 25)	RR (Out of a max score of 20)
Florida	Yes	346	17.17	18.48	15.14
	No	184	16.91	17.86	14.80
F			0.96	1.91	1.39
P			0.327	0.168	0.239
Gender	Male	132	16.7	17.8	14.6
	Female	398	17.2	18.4	15.1
F			2.90	1.85	2.74
P			0.089	0.174	0.098
Education	Less than high	16	16.13	18.38	14.56
	High school	126	16.35	16.98	14.18
	Some college	180	17.39	18.25	15.08
	College graduate	208	17.33	19.05	15.51
F			4.62	4.85	5.10
P			0.0034	0.0024	0.0017
Income2					
F			1.23	1.18	1.61
P			0.1507	0.2021	0.0097
Times2					
F			1.60	1.63	1.26
P			0.0593	0.0524	0.2127
Race	Black	95	16.21	17.60	14.23
	White	362	17.42	18.36	15.24
	Hispanic	52	16.62	18.90	14.96
	Other	21	16.38	18.00	15.00
F			5.56	0.96	2.70
P			0.0009	0.4118	0.0450

[1]In this table, the original respondents' scores of statements # 2, 3, 5, and 14 of Table 22 are reversed on the scale of 1 to 5 before being grouped with other statements in the respective category and the average value for the group is computed. [2]For brevity, only *F* statistic values are reported.

Table 24. Results of explanatory factor analysis

Variable	Factor 1 WRR	Factor 2 UWR
Florida's coral reefs have deteriorated dramatically in recent decades.	0.8064	-0.0972
I am comfortable with the level of risks facing Florida's coral reefs and marine resources.	0.8068	-0.0298
The health of Florida's coral reefs is managed by the relevant authorities.	.08388	0.0691
The risks to Florida's coral reefs and fisheries will continue to increase into the future.	0.8128	0.0240
Future generations will address the risks faced by Florida's reefs appropriately	0.7959	0.0576
Concern regarding the health of Florida's coral reefs	0.6741	-0.0704
Concern regarding overfishing in Florida and other US states/jurisdictions	0.2422	0.6044
Concern regarding marine pollution and loss of marine biodiversity	-0.1141	0.6065
Concern regarding rising ocean temperatures and bleaching of Florida's corals	0.5811	-0.0657
Concern regarding physical damage to coral reefs and sea grass beds.	-0.2428	0.5022
Government agencies must start to take actions to preserve and protect Florida's coral reef ecosystems.	0.7617	-0.0222
As a citizen, I am also responsible for contributing towards the protection and the enhancement of coral reefs.	0.7496	0.0367
Any human activities that adversely affect the health of coral reefs and fish populations should be regulated.	0.7387	-0.0213
The relevant public agencies will manage Florida's coral reefs without my contribution to the effort.	0.0698	0.5068

Loading on a given factor was assumed if loading >0.50 (shaded).

Table 25. Program cost combinations presented in survey

1	Status quo	0	9	Status quo	0
	Restocking	50		Restocking	50
	Marine reserves	40		Marine reserves	160
	Combined	85		Combined	200
2	Status quo	0	10	Status quo	0
	Restocking	50		Restocking	200
	Marine reserves	80		Marine reserves	40
	Combined	125		Combined	230
3	Status quo	0	11	Status quo	0
	Restocking	110		Restocking	110
	Marine reserves	40		Marine reserves	160
	Combined	140		Combined	255
4	Status quo	0	12	Status quo	0
	Restocking	110		Restocking	140
	Marine reserves	80		Marine reserves	120
	Combined	180		Combined	245
5	Status quo	0	13	Status quo	0
	Restocking	50		Restocking	200
	Marine reserves	120		Marine reserves	80
	Combined	160		Combined	265
6	Status quo	0	14	Status quo	0
	Restocking	140		Restocking	140
	Marine reserves	40		Marine reserves	160
	Combined	170		Combined	285
7	Status quo	0	15	Status quo	0
	Restocking	140		Restocking	200
	Marine reserves	80		Marine reserves	120
	Combined	210		Combined	305
8	Status quo	0	16	Status quo	0
	Restocking	110		Restocking	200
	Marine reserves	120		Marine reserves	160
	Combined	220		Combined	340

*Due to human error, cost combination six was not presented to respondents of the choice model questions presented in dichotomous choice format.

Table 26. Results of rank ordered logit

	Model 1	Model 2	Model 3	Model 4	Model 5	Model 6	Model 7
Bid	-0.0019**	-0.0019**	-0.0019**	-0.0019**	-0.0019**	-0.0018**	-0.0019**
	(0.0008)	(0.0008)	(0.0008)	(0.0008)	(0.0008)	(0.0008)	(0.0008)
Coral	0.3316	0.2446	0.2663	0.2581	0.3456	0.5487**	0.2179
	(0.4008)	(0.3890)	(0.3880)	(0.3654)	(0.3511)	(0.2838)	(0.2724)
Marine reserve	-0.1068	-0.1626	-0.1387	-0.0758	-0.0935	0.1908	-0.0140
	(0.3757)	(0.3644)	(0.3639)	(0.3403)	(0.3261)	(0.2629)	(0.2526)
Edu * coral	0.0676	0.0690	0.0728	0.0764	0.0721		
	(0.0738)	(0.0736)	(0.0733)	(0.0726)	(0.0725)		
Edu * MR	0.0959	0.0957	0.1026	0.0984	0.1006		
	(0.0696)	(0.0693)	(0.0691)	(0.0684)	(0.0683)		
Inc * coral	0.0043	0.0043	0.0045*	0.0044*	0.0046*	0.0052**	0.0058**
	(0.0027)	(0.0027)	(0.0027)	(0.0027)	(0.0027)	(0.0026)	(0.0025)
Inc * MR	-0.0011	-0.0011	-0.0008	-0.0005	-0.0006	0.0005	0.00031
	(0.0025)	(0.0025)	(0.0025)	(0.0025)	(0.0025)	(0.0024)	(0.0023)
Times * coral	0.0138	0.0129					
	(0.0182)	(0.0182)					
Times * MR	0.0219	0.0215					
	(0.0175)	(0.0174)					
Age * coral	0.0008	0.0005	0.0003				
	(0.0044)	(0.0044)	(0.0044)				
Age * MR	0.0014	0.0012	0.0009				
	(0.0042)	(0.0042)	(0.0042)				
Gender * coral	-0.3704***	-0.3721***	-0.3803***	-	-	-	-0.2332*
	(0.1399)	(0.1392)	(0.1386)	(0.1381)	(0.1379)	(0.1379)	(0.1323)
Gender * MR	-0.0189	-0.0168	-0.0283	-0.0402	-0.0413	-0.0413	0.0520
	(0.1317)	(0.1311)	(0.1308)	(0.1304)	(0.1301)	(0.1301)	(0.1259)
Flres * coral	0.0725	0.0985	0.1074	0.1075			
	(0.1272)	(0.1245)	(0.1238)	(0.1236)			
Flres * MR	-0.0621	-0.0437	-0.0277	-0.0242			
	(0.1209)	(0.1185)	(0.1178)	(0.1176)			
WRR * coral	0.5935***	0.5935***	0.5959***	0.5949***	0.5968***	0.6037***	
	(0.0701)	(0.0695)	(0.0695)	(0.0690)	(0.0690)	(0.0683)	
WRR * MR	0.4298***	0.4214***	0.4259***	0.4304***	0.4297***	0.4412***	
	(0.0652)	(0.0645)	(0.0644)	(0.0639)	(0.0639)	(0.0632)	
UWR* coral	0.2223***	0.2204***	0.2120***	0.2096***	0.2109***	0.2108***	
	(0.0771)	(0.0763)	(0.0754)	(0.0748)	(0.0747)	(0.0746)	
UWR* MR	0.2312***	0.2373***	0.2256***	0.2354***	0.2338***	0.2366***	
	(0.0731)	(0.0724)	(0.0717)	(0.0711)	(0.0711)	(0.0710)	
Enviro * coral	0.3419**	0.3575**	0.3680**	0.3706***	0.3745***	0.3688***	0.7374***
	(0.1473)	(0.1463)	(0.1454)	(0.1441)	(0.1440)	(0.1434)	(0.1326)
Enviro * MR	0.1355	0.1473	0.1643	0.1542	0.1545	0.1389	0.4181***
	(0.1402)	(0.1388)	(0.1380)	(0.1371)	(0.1371)	(0.1362)	(0.1266)
Dist * coral	-0.0001						
	(0.0001)						
Dist * MR	-0.0001						
	(0.0001)						
Observations	527	529	529	530	530	530	530
LR chi^2	222.93	224.47	222.46	223.41	222.61	211.60	67.95
Log-likelihood	-1563.371	-1568.955	-1569.961	-	-	-	-

Standard errors in parentheses; * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

Table 27. Annual HH WTP estimates

Program	Model 1	Model 2 Remove dist.	Model 3 Remove times	Model 4 Remove age	Model 5 Remove flres	Model 6 Remove edu	Model 7 Remove risk1, 2
Coral	119.11	95.97	100.76	98.25	141.05	179.01	94.72
Marine Reserves	14.81	.03	7.35	24.54	20.16	96.60	46.25
Both	133.92	96.00	108.11	122.79	120.89	275.61	140.82

Table 28. Marginal WTP results at various levels of risk perception

	Model 6 Attributes-Interacted with Risk Perception Model						
Risk Perception	$\overline{Risk} = 3.60$	1[a]	2[a]	3	4	5	Risk-weighted average WTP[b]
Coral	$179.01	*$0.00*	*$0.00*	$27.40	$270.35	$513.33	$155.27
MR	$96.60	*$0.00*	*$0.00*	*$0.00*	$183.72	$384.57	$222.05
Both	$275.61	*$0.00*	*$0.00*	$27.40	$454.07	$897.90	$377.30

[a]Computed WTP values were negative for risk perception levels of Likert scale 1, 2, and 3 (MR only). Since negative WTP values (disutility from improved attributes) seem unrealistic, those values were discarded and WTP values were assumed to be zero at risk perception levels of 1, 2, and 3 (MR only).

[b]Risk-weighted average WTP values are computed by using average percent of respondents expressing different levels (1 to 5) of agreement to all risk questions as weights. On an average, 5.73% strongly disagreed (Likert scale = 1), 10.78% somewhat disagreed (2), 28.54% neutral (3), 27.90% agreed (4), and 27.04% strongly agreed (5) to the fourteen risk questions.

Table 29. Aggregated WTP for restocking and combined programs

Program	Certified Florida divers	South FL HH	Florida HH	SE US HH	South FL coral users
Restocking	2,247,091	22,550,093	65,165,430	124,695,951	10,551,742
Combined	5,845,497	58,660,947	169,518,852	324,379,268	27,448,896

*5 hectares; 2017 dollars

5.0 Discussion and management implications

5.1 Nonmarket benefits from restocking and protecting staghorn corals

Staghorn corals are critical to the diversity and productivity of the FRT, supporting local and regional fisheries, tourism, recreation, and educational and spiritual experiences (Wilkinson, 2008; Principe *et al*., 2012). Staghorn coral was among most abundant and ecologically dominant corals on shallow Caribbean reefs for the last one million years until the 1970s and 1980s (Goreau 1959; Geister 1977; Adey 1978; Jackson 1992, 1994; Pandolfi 2002; Pandolfi and Jackson, 2001, 2006). Today staghorn corals in the Florida Keys occur primarily in patch reefs as opposed to their former abundance in deeper forereef habitats and, under current conditions, are believed to face localized extirpation in the next 100 years without active intervention (Miller *et al*., 2008); declines in abundance have been estimated at 97% in some locations. Active restoration to mitigate losses in coral cover is increasingly becoming considered a critical component of coral conservation and recovery efforts (Precht, 2006; Edwards and Gomez, 2007; Lirman and Schopmeyer, 2016; Schopmeyer, *et al*., 2017); Currently, tens of thousands of staghorn coral colonies are being transplanted annually onto Florida reefs. Nursery reared outplants are reaching sexual maturity within two years of outplanting and have been observed spawning, showing outplants can contribute to the species. Linking active restoration with other available management tools such as marine reserves is widely believed among practitioners to offer the highest likelihood of success to reef restoration efforts in areas impacted by human activities (Young *et al*., 2012). Changes in coral reef ecosystems and reef health will alter the life cycle of reef dependent fish species (Syms and Jones, 2000) and thus fisheries productivity, biodiversity, and economic value of the FRT.

5.2 Implications for coastal resource management

This study provides coastal resource managers with insight into the economic benefits of enhanced staghorn coral populations and overall coral reef ecosystem health on the FRT and addresses some of the recurring challenges of ecosystem restoration and management, including uncertainty regarding the existence and severity of risks and the need for intervention, ecological and economic benefits estimation from ecosystem restoration, and the appropriate distribution of costs in relation to the extent of benefits. Results of this study suggest the public believes the risks to Florida's coral reefs and fisheries will continue to increase in the future and that it is incumbent upon government agencies to take actions to preserve and protect Florida's coral reef ecosystems. Results also indicate the public is uncertain as to whether the relevant public agencies will manage Florida's coral reefs without their contribution and feels a responsibility to contribute to the protection and the enhancement of coral reefs, as evidenced by the substantial WTP estimates. The public's moderately high level of concern regarding the risks facing Florida's coral reefs and coastal resources may partially explain why respondents strongly supported the regulation of any human activities that adversely affect the health of coral reefs and fish populations.

As mentioned previously, the results of this study clearly indicate respondent risk characteristics influence their valuation of ecosystem services. Risk-adjusted and non-risk adjusted WTP values were estimated for comparison; At a risk level of five, the highest, WTP values for the marine reserve and both programs are substantially higher than non-risk-adjusted WTP values and WTP estimates for the three management alternatives from model 7, which contains no risk variables, average 97.86% lower than those from model 6, containing the risk variables; Inclusion of the two risk variables in the model approximately doubles WTP for each of the three alternatives confirming the magnitude of the influence of risk characteristics on WTP.

Valuation results are comparable with those of similar studies examining the public's values for coral reef and coastal ecosystem health suggesting broad support among the national population for the protection of coastal resources. Using a stated preference survey approaches Stefanski and Shimshack (2015) found WTP to expand marine protected areas in the northern Gulf of Mexico ranged from $35 - $107 per household and Bishop, *et al*. (2011) estimated mean WTP to implement marine reserves to protect 25% of the Hawaiian island's coral reef ecosystems to be $224.81, WTP to restore five acres of coral reefs annually to be $62.82.

Dichotomous choice and rank-ordered data are commonly fit using several different econometric models. Here, we assume the error terms are distributed extreme values and, accordingly, use conditional and rank ordered logit for the dichotomous choice and rank-ordered data, respectively. With the rank ordered logit, the probability of the respondents' second and third choices (conditional probabilities) in the choice model are the same as the unconditional probabilities, i.e., no statistical information about the respondent is gathered as the rank ordered logit fits the respondent's sequence of rankings (Train, 2002; Bishop, 2011). In practice, this means the choice model would perform just as well as a sequence of three separate choices made by three different respondents (Bishop, 2011). Employing an alternative econometric model like the rank-ordered probit, which does not treat respondent rankings as separate choices, may shed more light on the probability of various choice sequences among respondents.

An underlying objective of this study was to improve our understanding of the extent of the market for a large-scale coral restocking program in SE Florida through examination of the empirical relationship between household WTP and distance from the Florida Keys. The extent of the beneficiaries of, and market for, restoration efforts is a critical input in cost-benefit analysis of staghorn recovery efforts and estimation of project's net economic value. Further, knowledge of the extent of the market may help determine the appropriate scale of education and outreach efforts aimed at developing support for staghorn recovery as well as whether project costs should be borne at the county, state, or federal level, for example.

The insensitivity of household WTP to both distance from the Florida Keys and experience with coral reefs in the past three years suggests there may be something novel about the program, coral reefs, or staghorn corals that appeals broadly to coral reef users and non-users. One explanation may be staghorn's designation as threatened under the ESA. In a CV study examining the public's WTP to conserve endangered species, Samples *et al*., (1986) found that respondents allocated more of their conservation dollars to endangered but recoverable animals as compared with extremely common or extremely rare animals and, through a meta-analysis of 31 studies, Richardson and Loomis (2009) found that the non-market values of species in the US are sensitive to changes in the size of species population, suggesting WTP may be influenced by strategic considerations. Another explanation for the insensitivity of household WTP to distance may be that the public attributes value to the FRT's irreplaceability and uniqueness as the third largest barrier reef in the world and only barrier reef in North America.

Our findings of support for efforts to restock and protect staghorn corals among and users and non-users are in harmony with the listing of staghorn coral under the federal ESA and the leadership of NOAA, a federal agency, in implementing a regional restocking plan. Federal leadership suggests the FRT is considered an environmental amenity of national significance by the federal government and that as residents we all derive benefits from its presence and preservation.

Aggregated WTP values extrapolated to various relevant population are presented in Table 29. Relative to terrestrial private property values, the magnitude of several of the aggregated valuation estimates are substantial and may seem implausible. As Bishop *et al*., (2011) notes, comparison of the benefits from a hectare of terrestrial privately-owned property to the market and non-market benefits flowing from a hectare of coral reef ecosystem, a public good, is tempting but inappropriate according to economic theory, which distinguishes between private and public goods. Many of the benefits of staghorn restocking and protection are non-excludable and non-rival meaning no one can be excluded from the enjoyment of the passive use values generated by restocking and protecting staghorn corals, and one individual's enjoyment of those benefits does not impact others' enjoyment. The economic benefits from protection and restoration can, therefore, be much larger per unit area than would be true for private goods.

However, these extremely large values derived by extrapolating household WTP to state or regional populations may not translate into program support. Because non-use values often make up most of the total economic value of public goods like coral reefs, extrapolating to smaller populations, particularly users like scuba divers or tourists, for example, likely provides a more realistic estimation of values. Educating and targeting such user groups for financial and political support for regional conservation programs examined in this study may yield more favorable results.

The models presented here highlight the complexity of the determinants of public preferences and WTP for enhanced ecosystem services supported by staghorn corals. Socio-demographic and economic variables like age, education, and income were statistically insignificant in almost all the valuation models. The risk variables, WRR and UWR, however, were highly significant (at the 1% level) in every model. These results reveal that general concern about the health of Florida's coral reef ecosystems and perception of risks associated with the loss of staghorn coral populations play a prominent role in shaping consumer preferences for reductions in the risks facing Florida's coastal resources, with respect to the probability of participating in the market and WTP amount. The results of similar studies are mixed. For example, Alberini and Scasny (2010) found that risk characteristics, method for reducing risk, and income, drove most of the heterogeneity in respondent preferences while other individual characteristics (e.g., age and education) were less impactful; Hunter *et al*., (2012), however, found risk characteristics to be of secondary importance to individual respondent characteristics in influencing market participation and WTP. Nevertheless, the significance and magnitude of the coefficients of the WRR and UWR risk variables in this study suggest education and outreach could enhance support for the regional restocking program.

6.0 Conclusion

Results of this study suggest users and non-users associate substantial non-market benefits with the restoration and protection of staghorn corals and Florida's coral reef ecosystems that are not affected significantly by distance from the Florida Keys, where most of the active restoration in Florida is occurring. These results are relevant and timely for resource managers in SE Florida as staghorn restocking is scaled up regionally and appropriate sources of funding are considered. Also, of relevance for resource managers is the significant influence of risk perception, risk concern, and attitudes toward risk reduction actions on WTP. In the face of climate change and increasing threats to coral reef ecosystems, the public's perception of the condition of Florida's coral reefs, concern for future risk, and sense of personal responsibility will influence the level of political support for the restoration and protection of Florida's coral reef ecosystems. Programs to increase public awareness and literacy regarding the condition, threats, and outlook of Florida's staghorn corals and coral reef ecosystems may engender support and help ensure the persistence of regional staghorn populations.

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Chapter 3: Cost-benefit analysis of restocking staghorn coral on the Florida Reef

1.0 Introduction

Coral reef ecosystems on the Florida Reef Tract (FRT) provide critical habitat for thousands of species and recreational and spiritual opportunities for millions of people every year. Proximity to the Miami metropolitan area and Florida Keys has subjected the reef ecosystem to decades of intense human use, deteriorating water quality, coral bleaching and diseases, loss of living coral cover, and declining reef fish populations. Once among the most ecologically dominant structure building corals on reefs in the Caribbean and SW Atlantic, staghorn coral has declined in abundance an estimated 97% regionally since the 1970s (Goreau, 1959; Geister, 1977; Adey, 1978; Jackson, 1992; Pandolfi, 2002; Pandolfi and Jackson, 2001; NMFS, 2015). Today, staghorn corals occur as isolated colonies or fragments primarily on isolated patch reefs as opposed to their former abundance in deeper forereef habitats (Miller *et al*., 2008). Local fisheries have declined in productivity over the same period as the decline in staghorn coral abundance. Total commercial landings on the east coast of Florida dropped from 30,039 metric tons in 1980 to 9,769 in 2016, a reduction of 67.39%, and 23 of 35 species of groupers, snappers, hogfish, and grunts have been chronically over-fished since the 1970s according to National Marine Fishery Service (NMFS) standards (Ault, 1998).

Leeworthy and Bowker (1997) estimated 13.7 million visitor days, worth annual non-market use value of over $1.2 billion, are spent annually in the Florida Keys, 75% of which is derived from natural resource-based activities like snorkeling, scuba diving and fishing. The inextricable linkages between the economy and health of its coastal ecosystems and make management and protection of the Florida Keys' existing resources critical to the future of the island chain.

In response to the precipitous decline of regional populations and listing as "threatened" under the Endangered Species Act in 2006, the National Marine Fisheries Service (NMFS) formulated a recovery plan for the species (NMFS, 2015). Proposed recovery actions include propagating staghorn coral colonies in underwater nurseries and transplanting them onto denuded reefs along the FRT and establishment of no-take marine reserves to protect remaining natural and restocked populations. Both recovery actions are expected to increase sexual reproduction and support the long-term recovery of wild staghorn populations and their genetic diversity (NMFS, 2015). The abundance recovery criteria established in the recovery plan for staghorn coral (NMFS, 2015) is that thickets exist across approximately 5 percent of consolidated reef habitat in 5 to 20 m water depth within the fore reef zone; thickets are defined as either a) colonies $\geq$ 0.5 m diameter in size at a density of 1 colony per m^2 or b) live staghorn coral benthic cover of approximately 25 percent. Recovery of staghorn populations has been estimated to require 400 years at a cost exceeding $250,000,000 (NMFS, 2015).

Over the past decade, more than 100,000 staghorn colonies have been outplanted at over 100 sites on the FRT and approximately 50,000 staghorn colonies are expected to be transplanted annually over the foreseeable future. Outplanting capacity has been largely determined and limited by the availability of funding and achieving the recovery criteria established in the recovery plan will likely require substantial increases in annual outplant volume from current numbers. Positive changes in the structure and function of the coral reef ecosystem as outplants mature are expected to enhance recreational opportunities for recreational users and affect the population dynamics of most commercially harvestable reef fish species and, thus, fisheries productivity and revenue. Several studies have examined visitor preferences and the tourism and recreational value of coral reef habitat in the Caribbean and Florida Keys (e.g., Bhat, 2003), however, none have focused explicitly on the values supported by staghorn corals or attempted a cost-benefit analysis of restocking and protecting regional populations. Cost-benefit analysis can provide insights into the economic efficiency of management and regulatory actions; management or regulatory actions with benefits exceeding costs are considered economically efficient.

Using a bioeconomic model (Conrad, 1999) of a multi-stock fishery and stated preference valuation techniques, the first two chapters of this research attempt to apply the ecosystem service valuation process to monetize the value of restocking and protecting staghorn populations on the FRT considering two of the most important direct-use values supported by staghorn coral in the Florida Keys, commercial reef fish fishing and recreational diving. Specifically, this study forecasts and evaluates the change in the value of the selected ecosystem services between the future with restocking at current numbers (i.e., 50,000 outplants yr^{-1}), the future with restocking at current numbers and marine reserves protecting transplanted colonies (referred to hereafter as the “combined” program), and the future without restocking. The objective of this chapter is to synthesize the valuation results of the first two chapters and examine the business case for restocking and protecting staghorn corals on a large scale. Because we limit our analysis to “direct” use values, and do not consider “indirect” use or “non-use” values, this study represents a conservative cost-benefit analysis.

Cost-benefit analyses comparing the benefits of preserving or enhancing environmental resources with the opportunity costs for alternative decisions has become widely practiced over the past several decades and is recognized as the primary appraisal method for public investments and public policy (Farrow and Toman, 1998). An understanding of the multiple ecosystem service benefits and tradeoffs associated with staghorn restocking can support restoration efforts in several ways, including improving site selection and design, increasing stakeholder buy-in for restoration projects, enhancing the ability to leverage funding opportunities, and enabling the evaluation of the project in terms of economic efficiency. To our knowledge, this study represents a first attempt to incorporate simulated changes in staghorn abundance over time from recovery efforts into an ex-ante ES valuation framework.

2.0 Methodology

2.1 Theoretical Framework

The Millennium Ecosystem Assessment (MEA) defines "Ecosystem Services" (ES) as "the benefits people obtain from ecosystems" (MEA, 2005). Ecosystem services can be organized in terms of uses of value to human populations (Table 30), and examined in quantitative or qualitative terms, or through economic valuation. Economic valuation of ES attempts to identify the ways ES benefit humans and monetize these benefits for comparison to other sources of value to society (Principe *et al*., 2015) and is commonly used to support policy and decision makers in making investment and policy decisions (Waite *et al*., 2014).

Often, no formal markets exist for the goods and services provided by environmental resources, so their monetary values to people may not be readily observable. In such cases, a common approach to valuing changes in the quantity or quality of ES flowing from an environmental asset involves eliciting people's preferences for changes in the state of their environment. To estimate the *ex-ante* recreational diving value associated with restocking and protecting staghorn coral populations, we applied two attribute-based stated preference (SP) methods. Stated preference methods are commonly used in environmental valuation to gather data about respondent preferences for environmental amenities, typically through hypothetical scenarios presented in a survey format. Because SP preference techniques enable examination of public preferences for provision levels of goods or services that differ from levels observed currently or in the past, they are often the only approach available for providing the economic valuation inputs required for cost-benefit analysis. The results presented in this study were quantified in terms of the public's mean WTP (2017 $US) per hectare of rehabilitated coral reef with, and without, a marine reserve protecting restocked colonies.

To quantify the ecological and economic commercial reef fish fishery benefits supported by increased staghorn coral abundance, we modify a standard bioeconomic model of a multi-species fishery to allow for the influence of habitat on the commercial reef fish stock. A staghorn coral support function is included in the intertemporal bioeconomic harvesting problem through the growth function of the fish stock; impacts of a change in the support function were quantified in terms of changes in the long-run equilibrium conditions of the fishery with, and without, a marine reserve to protect transplanted staghorn colonies. This general methodology for quantifying staghorn-fishery linkages and the impacts of staghorn abundance and no-take marine reserves protecting rehabilitated reefs on the equilibrium conditions of the fishery can be applied regionally to staghorn restoration projects.

As mentioned previously, we limit our cost benefit analysis to the examination of two of the primary direct uses expected to benefit from enhanced staghorn abundance, commercial reef fish fishing and recreational diving. Accordingly, the valuation results presented here reflect only a partial accounting of the benefits anticipated from ongoing staghorn recovery efforts.

2.2 Estimating changes to ES values from restocking and protecting staghorn corals

2.2.1 Recreational Diving Value

To derive the contribution of recreational diving to the total value of staghorn restoration and protection, we administered a household survey to elicit the preferences and level of support of residents of the southeastern United States for restocking and protecting Florida's staghorn coral populations. The survey included a choice model that enabled estimation of the respondents' WTP for three hypothetical management alternatives. Residents of Florida, Alabama, Georgia, and Mississippi (n=3,135) were randomly selected to complete the survey; One-thousand and sixty-one surveys were completed and retained for analysis. In the survey instrument, each alternative is described in terms of its features or "attributes". Described attributes included: (i) the number of staghorn colonies outplanted on the FRT annually and estimated area of coral reef rehabilitated after 30 years of outplanting, (ii) the area of new marine reserves protecting outplanted corals, and (iii) cost of each alternative to the respondent. Attributes had two levels apiece: the status quo or some positive action. The status quo alternative consisted of the current level of outplanting (approximately 50,000 colonies yr^{-1}) continuing for at least 30 years with no new marine reserves to protect outplanted colonies; the positive actions consisted of: (1) increase staghorn outplants on the FRT from the current annual average of approximately 50,000 to 300,000, (2) implement no-take marine reserves to protect the 50,000 colonies currently outplanted every year, (3) increase staghorn outplants on the FRT from the current annual average of approximately 50,000 to 300,000 and implement no-take marine reserves to protect outplanted corals. For the combined program, existence of the marine reserve to protect outplanted corals was assumed to boost the intrinsic rate of growth of the commercially important reef fish stock from .2 to .3. The growth and area of outplanted colonies was simulated $A = \pi AB$, where A and B are one-half of the colony major and minor axis, respectively. Outcomes were characterized in terms of reef area rehabilitated with outplanted colonies upon reaching 54.31% coverage in the restoration site, the 95th percentile staghorn coverage estimated from an observational dataset of staghorn colony size and abundance and reef fish species, length and abundance collected in the Dry Tortugas from 2012-2014 (Miller and Huntington, 2015). Willingness-to-pay values derived from survey responses reflect the amount households were willing to pay, in 2017 dollars, for program outcomes. As with many public investments, the anticipated benefits of rehabilitating reefs with nursery-reared staghorn colonies will be realized at some future date, whereas most of the costs are incurred initially. Because the ecological value of newly outplanted staghorn colonies is negligible relative to their value upon maturity and full value is realized only upon reaching ecological equilibrium, we adjusted WTP values to account for the area of staghorn coverage at the restoration site as a percentage of the derived carrying capacity of 54.31%. To do this we follow the following steps:

(i) converted household WTP values to per-hectare WTP values

$$WTP_{ha} = \frac{WTP_{HH}}{area} \quad [34]$$

where WTP_{ha} is per the per-hectare WTP value, $WTP_{HH}$is household WTP derived from the rank order logit results presented in Table 25, and $area$ is the hectares of rehabilitated reef containing outplanted staghorn corals.

(ii) derived inflation-adjusted WTP values for years one through 30

$$WTP_t = (1 + r) * WTP_{t-1} \quad [35]$$

where WTP_t is per hectare WTP in period $t$and $r$is the mean rate of inflation in the US from 2009-2018, 1.65%; and

(iii) took the product of the ratio of staghorn coverage to carrying capacity and per-hectare WTP to arrive at an area, or coral growth-adjusted WTP value:

$$WTP_{adj,t} = WTP_t * \frac{cover_t}{K} \quad [36]$$

where $WTP_{adj,t}$ is the area adjusted WTP at time t, WTP_t is inflation adjusted per-hectare WTP at time t, $cover_t$ is the percent coral cover at the project site at time t, and K is the site carrying capacity of 54.31% staghorn coverage.

Extrapolating the adjusted per hectare WTP values for the alternative programs to the estimated population of certified open-water scuba divers in Florida (www.dema.org), we derived the contribution of recreational diving to the total economic value of staghorn recovery efforts.

Because divers are direct users of coral reefs whose consumer surplus has been shown to be enhanced by the health of the coral reef ecosystems they visit (Bhat, 2002), we felt like the recreational diving benefits supported by staghorn restocking represented the true project value, rather than a value derived from a larger population of users and non-users like the population of South Florida, for example.

2.2.2 Commercial reef-fish fishery value

We applied a deterministic bioeconomic model of a multispecies fishery (Conrad, 1999) that accounts explicitly for the effect of staghorn coral coverage on commercially harvestable reef fish biomass and productivity to quantify changes in the optimal equilibrium commercial reef fish stocks, harvest rate and profit from restocking and protecting staghorn coral populations. The model is spatially implicit, in that the precise relative location of each restoration site is not specified. Because stocks of the most economically important commercially harvested reef fish in Florida are managed, we examine equilibrium conditions characterizing maximum economic yield (MEY), or the stocks and harvest which maximize economic benefits to society, rather than that of an open-access fishery. Key parameters were estimated from existing datasets of regionally collected staghorn and reef fish size and abundance (Miller and Huntington, 2015; SEFSC, 2016). Bioeconomic model parameters requiring estimation included: (1) annual changes in rehabilitated reef area covered by outplanted staghorn colonies at the simulated restoration site resulting from restocking and protection, (2) baseline abundance of commercially important reef fish on the FRT inside and outside of areas prohibiting consumptive activities, (3) reef fish carrying capacity in the study area, (4) harvest cost, and; (5) the biophysical relationship between staghorn coral area and reef fish biomass.

To estimate the change in area of staghorn corals over time resulting from outplanting, we developed a simple linear staghorn growth model. For our baseline bioeconomic model run, the results presented here, we assume, at the time of outplanting, simulated colonies are elliptical in shape, 25 cm in length. Change in outplanted staghorn area was simulated following the equation for the area of an ellipse (Kiel, 2014),

$$Area = \pi AB \qquad [37]$$

where A and B are one-half the length and width of the colony's major and minor axis, respectively. Simulated colonies are assumed to be outplanted in a grid pattern at a uniform density of 10,000 outplants per hectare (ha) and assumed to maintain an annual major axis growth rate of 5 cm; published staghorn linear growth ranges from 3 to 11.5 cm yr^{-1} (Shinn 1966, Gladfelter *et al*. 1978).

We cap colony length at 100 cm (at which point colonies begin to interlock at the simulated treatment area and the marginal ecological value of continued growth declines) and cap coverage to 54.31% of the treatment area, the 95th percentile estimated from the Miller and Huntington (2015) dataset. Simulated outplants in the baseline scenario experience first and second year mortality of 15% and 10%, and none thereafter (Schopmeyer *et al*, 2017).

To derive baseline reef fish biomass and carrying capacity in the study area, we use an observational dataset of reef fish counts and measures inside and outside of no-take marine reserves in the FKNMS (SEFSC, 2016). We use the median biomass estimates as parameters representing fishery carrying capacity prior to restocking.

With double log-linear regression we quantify the biophysical relationship between staghorn coral coverage and reef fish biomass using a dataset of reef fish and staghorn colony measures and abundance collected between 2012-2014 from underwater visual surveys (n=65 transects) in the Dry Tortugas National Park, a relatively rich coral reef ecosystem at the western tip of the Florida Keys (Miller and Huntington, 2015)

Using the ex-ante estimates of outplanted staghorn coverage from the coral growth model, reef fish abundance and carrying capacity estimated from the SEFCS (2016) datasets, harvest costs derived from data queried by provided by professionals within the Fisheries Monitoring Branch (FMB) of the Southeast Fisheries Science Center (SEFSC) in Miami, Florida, market fish prices, and fish stock growth from peer reviewed literature (Froese and Pauly, 2018), and estimated diffusion coefficient, the model enables characterization of the linkages between coral abundance, commercial reef fish stocks and optimal sustainable harvest. A detailed solution of the bioeconomic model is contained in Chapter One.

3.0 Results and discussion

The monetized value of the subset of ecosystem services affected by restocking and protecting staghorn corals that are examined by this study are shown in Tables 31 and 32. In Table 31, household WTP results are presented for 300,000 25 cm^2 colonies outplanted annually for 30 years. The bioeconomic model annual WTP values were discounted at a 4% discount rate to arrive at the discounted NPV in Table 31 and reveal the incremental benefit of management alternatives (restocking and protection) over no staghorn restocking. Table 33 presents mean household WTP results for a one-time planting of 50,000 25cm^2 staghorn colonies under each of the two management alternatives, extrapolated to the mean population of certified open water scuba divers in

Florida over the past three years. Results confirm that recreational values are dramatically larger than commercial reef fish fishery values.

The adjusted per hectare annual WTP values are presented in Table 34. Because outplanted colonies do not reach their carrying capacity until year 22, adjusted values are less than base values for years 1-22 and reach base values in year 22, at which staghorn coverage reaches its assumed carrying capacity of 54.31% of the restocked reef (Figure 8). Project net present values for 50,000 staghorn colonies outplanted annually for 30 years are presented in Table 35. Values were derived by extrapolating adjusted WTP values to the various relevant populations and accounting for costs of production, outplanting, and two years of monitoring (Coral Restoration Foundation, personal communication). Corresponding benefit-cost ratios and sensitivity analysis are presented in Table 36. Results suggest project values may be substantial, and benefit-cost ratios may be greater than one, suggesting economic efficiency, depending on the relevant population considered. A description of all the ecosystem services supported by staghorn corals is contained in Table 29. As noted previously, the services valued with this study represent a subsample of those supported by staghorn corals.

4.0 Management implications and conclusions

The work presented here is consistent with previous research revealing that rehabilitation and restoration of ecosystems and the goods and services provided can yield significant contributions to society and to economies, and highlights some of the key challenges when attempting to monetize the value of ecosystem services. This study focuses on two of the most important direct use values supported by increased staghorn coral abundance in the Florida Keys, commercial reef fish fishing and recreational diving.

Because we examine only a subset of the ecosystem services likely to be impacted by the recovery of staghorn populations, the benefits highlighted here represent only a partial accounting of the total economic value of enhancing local populations; The economic value of several key ecosystem goods and services supported by the recovery of staghorn corals off SE Florida remain unexamined. Previous work has reported that non-use values make up a substantial portion of the TEV of coral reefs, suggesting that the values reported here provide only a small fraction of the total value of efforts to recover lost staghorn populations.

Future research examining the impact of staghorn restocking on the recreational fishing industry and the provision of physical coastal protection, for example, could fill some of the remaining gaps in our understanding of the total economic value of coral restoration efforts. More than half of the economy of the Keys is supported by ocean recreation and tourism. Given the strong economic linkages between marine ecosystem health and the rest of the economy, an *ex-ante* input-output analysis accounting for multiplier effects in the local and regional economies impacted by staghorn restocking may shed more light on the net total economic value of recovering lost staghorn populations as well as contributions to individual sectors of the Florida Keys economy.

Coastal ecosystems provide a substantial proportion of the population of Florida physical protection from the impacts of strong tropical storms. An examination of the contribution of staghorn restocking to the coastal protection value of the FRT can also further our understanding of the TEV of recovering staghorn populations. As sea-level rise continues, the intensity of tropical storms continues to increase with climate change

(Bender *et al*., 2010), and the population of SE Florida continues to grow, the coastal protection value of staghorn recovery efforts will likely increase.

Although implementation of a comprehensive regional restocking program and marine reserves to protect staghorn populations are two of the recovery actions identified in the species recovery plan (NMFS, 2015), the management alternatives examined in this study were not based on actual proposals. Our hypothetical scenarios are simply tools to estimate the total value of restocking and protecting staghorn corals. Actual efforts to rehabilitate denuded reefs and implement new marine reserves on a large-scale in the Florida Keys will likely face various obstacles, including major gaps in reef restoration science, and social and institutional inertial resistance to change (Bohnsack, 1999).

Results of our household survey suggest a substantial percentage of the public supports efforts to enhance and protect staghorn coral populations off SE Florida. The perceived benefits of healthier coral reef ecosystems are generally positive and potential dis-benefits from coral gardening and restocking denuded reefs are considered negligible. Even with broad public support for the proposed interventions, however, distributional issues associated with marine reserves will likely result in resistance from special interests (Bohnsack, 1999). For example, extractive users may oppose restrictions prohibiting fishing on restocked reefs over concerns of potential congestion on the remaining fishing grounds, increased fuel costs, or user conflicts. A general distrust of science and management among users has impeded past efforts to establish new areas closed to consumptive activities in SE Florida (Seeteram, *et al*., submitted for publication) and may inhibit the implementation of new no-take marine reserves for the protection of restocked corals. Although no-take marine reserves have proven effective in the Florida Keys in enhancing biomass of harvested species within reserve boundaries, and research suggests diffusion of fish moving out of marine reserves may improve fishing, many would prefer a marine reserve anywhere other than where they traditionally fish. (Seeteram, *et al*., submitted for publication) found that 66.7% of surveyed commercial fishermen opposed expansion of no-take zones in the FKNMS due to their perception that proposed changes would hinder their current fishing operations and 23.3% of those surveyed who opposed expansion did so even if the proposed management action would not hurt their business. These findings suggest the effectiveness of efforts to recover local staghorn populations will likely depend on the ability of managers to influence the perception of local users.

Our baseline model assumption that implementation of a marine reserve would not enhance outplanted survivorship may undervalue marine reserves as a tool for staghorn and fisheries conservation. Research suggests no-take marine reserves protecting corals from damage associated with fishing gear, anchoring and other physical stressors may enhance coral survivorship, recruitment and growth (Mumby *et al*., 2007), however, the effectiveness of marine reserves in preserving outplanted corals in the Florida Keys has not been established. A global meta-analysis examining the effectiveness of marine reserves in protecting coral reefs found, on average, no change in coral cover on reefs protected by coral cover while reefs outside of marine reserves experienced losses in coral cover, on average (Selig and Bruno, 2010). Mumby, *et al* (2007) reported denuded coral reefs in the Caribbean recovered four times faster when protected by marine reserves. Variation in recovery times between protected and unprotected reefs examined as part of that study was attributed primarily to reductions in macroalgae cover in marine reserves from rebounding stocks of overharvested parrotfish, whereas coral cover loss off SE Florida has been attributed primarily to disease, with which marine reserves will have little to no direct impact.

Our assumptions of linear coral growth and spatial homogeneity across the area rehabilitated through outplanting are not realistic. As thickets develop, there are years of healthy growth or die off, which may be the result of disease, predation, storms, or other environmental factors, for example. Staghorn's primary mode of reproduction is through asexual fragmentation and nursery-reared colonies have been observed reproducing sexually within two years of outplanting. Because the capability to predict the contribution of reproduction to the rate of change in outplanted staghorn cover is limited, and to minimize the likelihood of overestimating changes in staghorn abundance over time in our modeling, we did not account for the contribution of reproduction, either sexual or asexual, in our coral growth model. We assume that any potential overestimation in outplant cover over time resulting from our assumptions of linear growth and spatial homogeneity will be offset by the omission of reproduction from our outplant growth simulation.

Our current capability to reliably value changes in the services and benefits flowing from restocked reefs is limited by major gaps in reef restoration science, including knowledge of critical physical and biological linkages. As our understanding of these linkages improves, our ability to more accurately characterize the relationships between staghorn abundance, the ecological functioning of coral reef ecosystems, and economic systems will also improve.

Large-scale rehabilitation of denuded coral reef habitats is now widely considered the only hope for recovery of the coastal fisheries, biodiversity, and shoreline protection that only large healthy reefs can provide. This study represents a first step in developing a reliable valuation framework for evaluating two of the most important direct-use values affected by coral reef rehabilitation. As the science of coral reef restoration evolves and more long-term data documenting the outcomes of individual projects becomes available, some of the uncertainty endemic to this study may be reduceable. While focusing on improving our ability to enhance the structure and function of coral reef ecosystems on the FRT, the success of restoration efforts will likely depend on addressing the needs of relevant stakeholders who are often the most direct recipients of ecosystem services. By putting actual estimates of costs and benefits to restoration projects, valuation studies like this one can help inform decisions related to sustainable resource use and management.

Table 30. Staghorn coral contributions to communities of the Caribbean region

Direct extractive uses	**Direct non-extractive uses**
Commercial fishing Recreational fishing Aquarium trade	Recreation (i.e., scuba diving, snorkeling, boating)
Indirect uses	**Nonuse values**
Essential habitat for associated reef species Reef building/framework construction Carbonate deposition Topographical relief/complexity Protection from wave action/erosion Biodiversity Microhabitat diversity	Aesthetics Scientific Value Educational Value

Adopted from: (Bruckner, 2002; Principe *et al*., 2015)

Table 31. HH WTP for alternative programs estimated using rank-ordered logit

Program	HH WTP
Coral	179.01
Marine reserve	96.60
Both	275.61

Table 32. Bioeconomic model results: 5 ha treatment

Management alternative	Discounted revenue stream	Per hectare value
No Restocking	$6,377	$425.13
Coral Restocking	$6,509	$433.94
Combined Program	$7,788	$519.23

One-time planting of 50,000 colonies after 30 years, 15 ha fishery

Table 33. WTP for one-time planting of 50,000 colonies (5 ha) extrapolated to certified open water divers in Florida

Year	Coral Restocking Program	Combined Program
1	$10,445	$26,971
2	$13,534	$37,407
3	$18,763	$51,861
4	$24,944	$68,945
5	$32,117	$88,770
6	$40,324	$111,454
7	$49,608	$137,115
8	$60,014	$165,878
9	$71,588	$197,867
10	$84,377	$233,215
11	$98,429	$272,056
12	$113,795	$314,527
13	$130,527	$360,772
14	$148,676	$410,937
15	$168,299	$465,173
16	$189,451	$523,635
17	$202,085	$558,556
18	$215,067	$594,438
19	$228,405	$608,483
20	$242,107	$617,975
21	$256,181	$627,616
22	$261,361	$637,406
23	$265,438	$647,350
24	$269,579	$657,449
25	$273,785	$667,705
26	$278,056	$678,121
27	$282,393	$688,700
28	$286,799	$699,443
29	$291,273	$710,355
30	$295,816	$721,436

Table 34. Adjusted and unadjusted annual WTP

Year	Unadjusted WTP ha^{-1} (2017 $)	Coral Index	Adjusted WTP ha^{-1} (2017 $)
0			
1	0.17	5.53	0.01
2	0.18	7.06	0.01
3	0.18	9.63	0.02
4	0.18	12.61	0.02
5	0.18	15.99	0.03
6	0.19	19.76	0.04
7	0.19	23.94	0.05
8	0.19	28.52	0.05
9	0.20	33.50	0.07
10	0.20	38.87	0.08
11	0.20	44.65	0.09
12	0.20	50.83	0.10
13	0.21	57.41	0.12
14	0.21	64.38	0.14
15	0.21	71.76	0.15
16	0.22	79.54	0.17
17	0.22	83.54	0.18
18	0.22	87.54	0.20
19	0.23	91.54	0.21
20	0.23	95.55	0.22
21	0.24	99.55	0.23
22	0.24	100.00	0.24
23	0.24	100.00	0.24
24	0.25	100.00	0.25
25	0.25	100.00	0.25
26	0.25	100.00	0.25
27	0.26	100.00	0.26
28	0.26	100.00	0.26
29	0.27	100.00	0.27
30	0.27	100.00	0.27

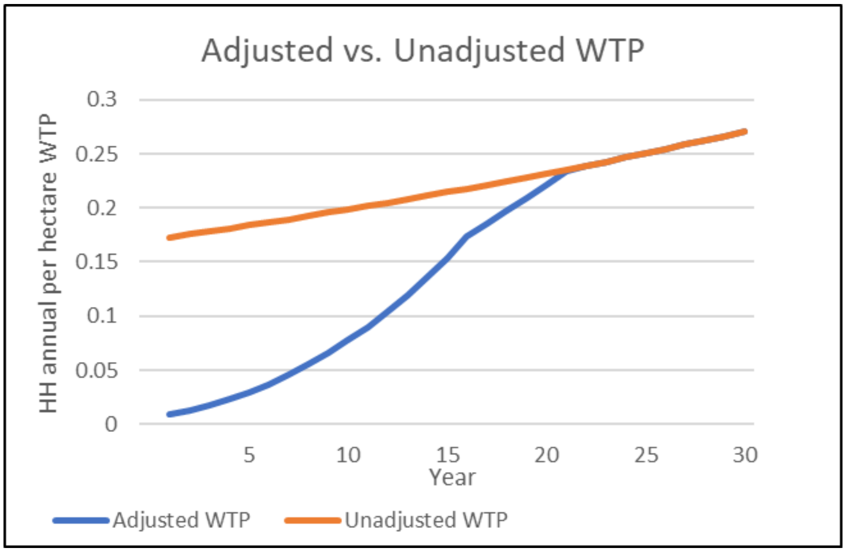

Figure 8. Adjusted. vs. unadjusted WTP

Table 35. Discounted NPV for 5 hectares annually for 30 years aggregated to various populations

Program	Certified Florida divers	South FL HH	Florida HH	SE US HH	South FL coral users
Restocking	2,247,091	22,550,093	65,165,430	124,695,951	10,551,742
Combined	5,845,497	58,660,947	169,518,852	324,379,268	27,448,896

Table 36. Benefit-cost ratios: 5 hectares annually for 30 years

Scenario	Florida divers	South FL HH	Florida HH	SE US HH	South Florida Coral
Restocking	0.66	6.66	19.25	36.84	3.12
100% increase in restoration costs	0.33	3.33	9.63	18.42	1.56
100% increase in r	0.34	3.44	9.94	19.01	1.61
Combined	1.73	17.33	50.09	95.84	8.11

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