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Essays on Economic Valuation of Water Resources

Christina Estela Brown
Florida International University, cbrow031@fiu.edu

FLORIDA INTERNATIONAL UNIVERSITY

Miami, Florida

ESSAYS ON ECONOMIC VALUATION OF WATER RESOURCES

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Christina Estela Brown

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

This dissertation, written by Christina Estela Brown, and entitled Essays on Economic Valuation of Water Resources, 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.

Michael Sukop

Jennifer Rehage

Pallab Mozumder

B M Golam Kibria

Mahadev Bhat, Major Professor

Date of Defense: June 20, 2019

The dissertation of Christina Estela Brown is approved.

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

Andrés G. Gil
Vice President for Research and Economic Development
and Dean of the University Graduate School

Florida International University, 2019

DEDICATION

This dissertation is dedicated to my family, my brilliant and supportive wife, Elizabeth, and our cat, Dax.

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ABSTRACT OF THE DISSERTATION

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Christina Estela Brown

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Professor Mahadev Bhat, Major Professor

Increased potential of flooding caused by heavy precipitation events and sea level rise, as well as growing risk of drought that are likely changes in the frequency and spatial distribution of climatic conditions, pose particular challenges to water management in coastal areas. Extreme events are expected to increase the complexity of managing scarce water resources for competing water users. South Florida, which is characterized by a mosaic of urban settlements, agricultural areas and natural areas, is served by a highly human-engineered water management system grappling to meet multiple objectives, including urban and agricultural water supply, flood control, and environmental restoration. Climate-induced water shortage or excess often tests the limits of the water management engineering system. While the Everglades suffers a lack of freshwater inflows, heavy precipitation and flooding events in the U.S. and worldwide in recent years have greatly damaged crop production. If model projections of increased weather extremes are realized, the cost of crop losses could increase drastically. These costs may be borne directly by the farmers impacted or transferred to private insurers or governmental disaster relief programs. The present research quantifies monetary values of lost recreational fishery ecosystem services due to reduced freshwater flow in the

Everglades using a survey-based discrete choice methodology, estimated at over $25 million annually. Examining survey respondents' willingness to pay for ecosystem services in light of their perceptions and preferences regarding the risks posed by climate change and sea level rise, when willingness to pay values were adjusted for risk perception the annual overall ecosystem service valuation (benefit) of users was 40.03% higher than the annual benefits estimated using non-adjusted willingness to pay. The economic value of crop flooding indemnity claims is also be estimated at a county level using a Stackelberg game-theoretic model, finding that in many years total indemnity far exceeds premiums, which are set at levels below farmers' maximum willingness to pay.

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1. Introduction

The Intergovernmental Panel on Climate Change (IPCC) report outlines changes in the frequency, spatial distribution, and magnitude of several climatic conditions and extreme events that are likely to occur in the not too distant future that could pose significant risks to human well-being (IPCC, 2014). Among such changes are an increased potential of flooding caused by increased heavy precipitation events and sea level rise, as well as increased risk of drought, and pose particular concern to coastal communities and agricultural production. South Florida is among the areas of the U.S. most vulnerable to inundation (Dolan & Walker, 2006; Erwin, 2009; Gornall et al., 2010; Scavia et al., 2002). In addition to inundation, rising sea level can cause shoreline erosion and inland migration, and increase salinity of freshwater ecosystems and aquifers (Scavia et al., 2002). These extreme events are in turn expected to further increase the complexity of managing scarce water resources for competing water users. Specifically South Florida, which is characterized by a mosaic of urban settlements, agricultural areas and natural areas, is served by a highly human-engineered water management system (Harwell et al., 1996). Management agencies grapple with managing water to meet multiple objectives, including urban and agricultural water supply, flood control, and environmental restoration. Climate-induced (flood, drought, sea level rise) water shortage or excess often tests the limits of this engineering system.

The Everglades National Park, including Whitewater Bay, Tarpon Bay, and Florida Bay, is renowned for its world-class recreational fisheries, generating more than $1.2 billion in annual economic activity (Fedler, 2009). The timing, quantity, and quality of freshwater inflows can greatly affect salinity and water quality regimes in south Florida coastal bays.

Freshwater flows are a key determinant of habitat and fisheries resource productivity (Rudnick, Ortner, Browder, & Davis, 2005; Stabenau, Engel, Sadle, & Pearlstine, 2011; Walters, Gunderson, & Holling, 1992), making the recreational fishing industry in the area a direct beneficiary of improved and sustained fishery habitat.

Historically, water flowed south from Lake Okeechobee into a broad, slow moving shallow river of water. At present, these flows are constrained by a dike and levy system and occupy less than half of their original areal extent, relegating the Everglades to part of a complex watershed management system regulated primarily for agriculture, flood control, and consumptive uses (Ogden, Davis, Jacobs, Barnes, & Fling, 2005; F. H. Sklar, Fitz, Wu, Van Zee, & McVoy, 2001; Sklar et al., 2005). As a result, the flow of freshwater through the Everglades has been reduced, channelized, and otherwise modified such that salinity regimes, biota, and a variety of ecosystem services in the coastal Everglades have dramatically changed (Perry, 2008; Rand & Bachman, 2008).

While the Everglades suffers a lack of freshwater inflows, heavy precipitation and flooding events in the United States and worldwide in recent years have greatly damaged crop production. If model projections of increased weather extremes are realized (IPCC, 2014; National Park Service, 2009), the cost of crop losses could increase drastically. Recent studies have attempted to simulate the effect of plant damage from excess soil moisture in order to estimate crop production loss, finding that losses under current climatic conditions may double in the next thirty years to an estimated $3 billion annually (Rosenzweig, Tubiello, Goldberg, Mills, & Bloomfield, 2002). These costs may be borne directly by the farmers impacted or transferred to private insurers or governmental disaster relief programs.

The following research will quantify monetary values of lost recreational fishery ecosystem services caused by reduced freshwater flow in the Everglades, as well as the economic value of crop flooding indemnity claims at a regional or county level. With the risk of future losses increasing because of the uncertainty of extreme events and the effects of climate change and sea level rise, quantifying these values is essential to evaluating potential policy or management responses. Ultimately, my research will use these estimated economic values within a penalty function framework to simulate potential climate or policy scenarios.

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2. Ecological-Economic Assessment of the Effects of Freshwater Flow in the Florida Everglades on Recreational Fisheries

2.1 Introduction

Everglades National Park (ENP), at the southern end of the Florida peninsula at 1.5 million acres, comprises the largest subtropical upland to marine ecosystem in North America. Everglades National Park contains a range of freshwater sloughs, seasonally flooded marl prairies, tropical hardwood hammocks, pine rocklands, and mangrove and seagrass-dominated estuarine habitats (Gunderson, 1994; Richardson, 2010; Saha et al., 2012). The Everglades, as an important migratory corridor, provides breeding and foraging habitats for over 400 species of birds, but also water storage and recharge for the Biscayne aquifer, the principal source of freshwater for regional human consumption (Lorenz, 2014; Saha et al., 2012).

South Florida's regional ecosystem is characterized by two distinct seasons, a wet season (generally from May-October) and a dry season (generally from November-April) (Saha et al., 2012; Brandt et al., 2012). While the average annual rainfall exceeds 60 inches, variation in tropical weather systems may result in wide seasonal variation and large year-to-year fluctuations (1901-2000 standard deviation of 11 inches in the Miami-Dade area) (Abetew and Huebner, 2001; National Park Service, 2009). Brandt et al. (2012) report that approximately 77% of the total annual rainfall occurs during the wet season, and remaining 23% during the dry season.

Prior to the development of the large freshwater drainage system in South Florida in the early and mid-20th century, water flowed south from Lake Okeechobee into a broad, slow-moving, shallow river of water. In the post-development period, these flows are constrained by a dike and levy system and occupy less than half of their original areal extent, relegating the Everglades to part of a complex watershed management system regulated primarily for agriculture, flood control, and consumptive uses (Ogden et al., 2005; Sklar et al., 2001, 2005). As a result, the flow of freshwater through ENP has been reduced, diverted, channelized and otherwise modified such that salinity regimes, biota, and a variety of ecosystem services in the coastal Everglades have dramatically changed (Perry, 2008; Rand & Bachman, 2008).

As a large, subtropical estuary averaging in depth from 6 to 9 feet, Florida Bay provides critical habitat for a variety of species, including seagrasses and coastal mangrove communities (Bachman & Rand, 2008). It serves as a nursery for larvae and juveniles of many critical species, including fish and wading birds (Lorenz, 2014).

The ENP, encompassing Whitewater Bay, Tarpon Bay, and Florida Bay, is renowned for its world-class recreational fisheries. Commercial fishing has been banned in Park waters since the 1980s. Recreational fishing in the greater Everglades area generates more than $1.2 billion in annual economic activity, with largemouth bass, red drum, snook, Atlantic tarpon, gray snapper and bonefish providing the largest economic impact (Fedler, 2009). Timing, quantity, and quality of freshwater inflows can greatly affect salinity and water quality regimes in south Florida coastal bays (Wang et al. 2003). Freshwater flows are a key determinant of habitat and fisheries resource productivity (Rudnick et al., 2005; Stabenau et al., 2011; Walters et al., 1992), making the recreational fishing industry in the area a direct beneficiary of improved and sustained fishery habitat.

Surface water stage (water depth relative to a given datum) and salinity gradients are strongly influenced by the amount of freshwater released through water management structures along the northern boundary of ENP (Stabenau et al., 2011; Childers and Leonard, 2005). These flows are regulated by the South Florida Water Management District (SFWMD) through massive canals and water-control structures. The SFWMD determines monthly water delivery targets for the Everglades wetlands on the basis of the historical water flow levels (South Florida Water Management District, 2014). However, in the recent years, average monthly deliveries have fallen short of these regulatory flow targets by more than 80% in some months. Managers are interested in understanding the potential ecological and economic impacts associated with water deliveries relative to the pressing demands of non-environmental sectors (e.g., agriculture, urban needs, etc.).

The goal of our paper was to develop a systems approach to systematically measure the economic impacts to changes in Everglades recreational ecosystem services relative to changes in freshwater management. We developed an integrated ecological-economic methodology by linking the Everglades hydrology to fisheries production and then modeled the effects of freshwater flows on several robust biological indicators. We quantified various attributes of the recreational fishing experience, and, finally, link the hydrology-influenced anglers' fishing experience to economic values.

Following Johnston et al. (2011; 2012), economic values are developed using a stated preference discrete choice experiment, taking care to provide respondents with the relevant ecological and hydrological knowledge essential for making informed choices to ensure valid willingness to pay estimates. At the end, this integrated methodology allows us to estimate losses in economic welfare caused by missing monthly freshwater delivery targets in the Everglades. These welfare losses are simply the foregone benefit or penalty of failing to meet exogenously determined freshwater flow targets. These penalty estimates serve as useful decision-support metrics for water resource managers making regional water resource allocations. While the conceptual model of the penalty function has been used in hydro-economic optimization (Harou et al., 2009; Jenkins et al., 2004; Newlin et al., 2002), its application to ecosystem services in terms of recreational fisheries is novel. In particular, the flexibility of the penalty function approach lends itself to applications to management scenario analysis and evaluation of potential restoration projects. This study advances ecosystem services valuation methods through its integrated hydrological-ecological-economic model.

2.2 Methods

2.2.1 Delineation of the Study Area

The geographic focus of the study is the ENP watershed, in particular the Shark River Slough (SRS) (Figure 1). Our goal is to assess the economic value of managing water through the Northern boundary of ENP. The relevant water structures involved in these flows are S12A-D, S333, and S334, located along Tamiami Trail (U.S. 41) at the northern boundary of ENP. The SRS region is bounded by state road U.S. 41 to the north, Gulf of Mexico to the southwest, Miami Rock Ridge to the east, and marl prairies to the west. The areal extent of the slough considered in this study is approximately 1700 km^2 (Saha et. al., 2012). At the western end of the slough is an estuarine zone including mangrove forests that extends approximately 30 km inland from the Gulf of Mexico. On the northern end, a ridge and slough landscape dominates, with sawgrass marshes and tree islands along the ridges, and floating and submerged aquatic macrophytes in the sloughs (Saha et. al., 2012; Price, 2008).

The majority of the inflow going through the above hydrological structure and into the ENP (70%) flows through Shark River Slough, with the remaining inflows reaching Taylor Slough to the southeast (Price, 2008). More than 90% of the flow through SRS region discharges into the Gulf of Mexico through five major rivers along the southwest coast (Levesque, 2004), corresponding to zones 4, 5, and 6 of ENP (Figure 1). Lostmans River contributes 33% of mean annual discharge, Harney River 32%, Broad River 17%, Shark River 14%, and North River 3%. While salinity fluctuates seasonally, there is an observed salinity gradient with Lostmans River at the north being saline and North River at the south being brackish (Woods, 1994).

The region's climate is seasonal subtropical, with wet and dry seasons, and it rarely experiences freezing temperatures. The dry season is November through April (Price et al., 2008; Saha et al., 2012), during which some parts of the slough are dry. Average water depth during the wet season of May through October is 1 m in the northern extent, and increases to about 3 m in the channels draining into the Gulf of Mexico (Saha et al., 2012).

2.2.2 Conceptual Model

Figure 2 is a schematic representation of our integrated model that captures the relationship between the freshwater flow and the periodic total monetary value of recreational ecosystem services enjoyed by anglers. The model first recognizes that freshwater discharges that flow into the coastal creeks are a key determinant of the overall health of the ecosystem in general and the fishery habitat in particular. Thus, the key indicators of the Everglades natural habitat quality including stage, primary fishery productivity, diversity, and location of fish depend on the freshwater flows (Higman, 1967). The model then recognizes that anglers who fish in ENP value various fishery and non-fishery attributes as part of their fishing experience, including catch per effort and enjoying a healthy natural area. That is, the overall recreational value of a fishing trip to ENP is assumed to be comprised of multiple attributes of anglers' experience: fishing-specific attributes (catch rate, size of the largest keeper, fishing travel time, etc.) and experiencing a healthy ecosystem (Johnston et al., 2012). Finally, the model monetizes the average individual fishing experience by using their mean willingness to pay as a proxy for their recreational value and then extrapolates the same to the entire population of anglers. The final stage of the modeling is to develop an aggregate penalty function that captures the recreational ecosystem values lost as a result of maintaining periodic water flows below the targets. The following sub-sections explain various hydrological, ecological, and economic sub-components of the model.

Hydro-ecological models

We first developed models that link hydrology with fishery productivity and overall ecosystem health. We linked the fishery catches with the managed S12 structures flow in two steps: (i) fish productivity in SRS coastal estuaries was assumed to be a function of SRS freshwater outflow into coastal streams and season (see equation 1 below) (Rudnick et al., 2005; Stabenau et al., 2011; Walters et al., 1992); and (ii) freshwater outflow was modeled as a function of S12 managed flow along with other hydrological variables related to the SRS watershed (see equation 2 below) (Saha et al., 2012). That is, the managed flow at the northern boundary of the SRS watershed indirectly affects the fish catches in the coastal areas through its effect on the freshwater outflows.

Following Rudnick et al. (2005) and Stabenau et al. (2011), we assumed that natural freshwater outflows into the coastal creeks and overall climatic conditions represented by the season were the key determinants of fish productivity. We recognize that the relationship between fish catch and freshwater flow is complex. While the freshwater flow could affect the distribution of certain species, and in turn, its catch, the anglers that are loyal to that species may follow those fish by changing their fishing location, traveling longer distance, and/or spending more time fishing. As a result, they may not see a fall in the amount of actual catch in relation to freshwater flow. Unfortunately, historical data on anglers' response in terms fishing location and travel distance appear to be unavailable. We partially address the data problem by defining fish productivity by CPUE, a measure of how many fish an angler caught per hour of fishing time, whether it was kept or not. In response to reduced freshwater flow, if anglers had to travel greater distances or spend more time to acquire a target amount of catch, the corresponding catch per unit effort (fishing time) would be lower than usual.

The CPUE is calculated for each of the following five species: Snook (*Centropomus undecimalis*), Red Drum (*Sciaenops ocellatus*), Tarpon (*Megalops atlantica*), Gray Snapper (*Lutjanus griseus*), and Spotted Seatrout (*Cynoscion nebulosus*). These five species were selected after consultation with ecologists and were also among the top species targeted by anglers surveyed (see subsequent sections for anglers' survey). We considered fishery productivity for the ENP fishing areas north of Flamingo and south of Chokoloskee, comprising zones 4, 5, 6S, 6C, and 6N. These zones include

Whitewater Bay, Shark River, Harney River, Broad River, Tarpon Bay, and Lostmans River.

$$C_m = a_{11}O_m + a_{12}S_1 + a_{13}S_2 + a_{14}S_3 + \varepsilon_1, \qquad (1)$$

where C_m is the catch in numbers of fish per unit of fishing effort in month m; O is the total surface water outflow from the SRS watershed to the southwest ENP coastal tributaries (KAF);

The variables S_1, S_2, and S_3 are the dummy variables representing the four seasons of the year (Winter, Spring, Summer, and Fall). As the model used time series data, the error term was expected to be auto-correlated. Notice that equation (1) is a simple additive model linking fish catches with management-induced freshwater outflows of the SRS estuaries. Alternative statistical relationships including logistic, double-log, saturation function, and quadratic forms did not fit the data as well as the linear model. One possible reason the logistic or other non-linear models were not a good fit was that, except during a handful of months, the flows during the model study period (1991-2005) were far from the "natural" flow targets.

Saha et al. (2012) computed SRS daily water surplus as a net effect of inflows, precipitation, and surface water losses caused by outflows, percolation, seepage, and evapotranspiration. The SFWMD (2005) also uses a similar daily water balance equation to simulate various monthly surface and ground water inputs and outputs. The purpose of our analysis was to link the SRS surface water outflow along western boundary (O) with the SRS surface inflows along the northern boundary. Childers and Leonard (2005) opine that the freshwater inflow through the S12 structures is the dominant factor that influences the freshwater discharges into the SRS coastal tributaries. Slightly modifying the water balance equations in Saha et al. (2012) and SFWMD (2005), we adapted the following simplified hydrological equation to link coastal freshwater outflow with the managed inflow of freshwater along the SRS northern boundary,

$$O_m = a_{21}F_{m-1} + a_{22}R_{m-1} + a_{23}L_{m-1} + \varepsilon_2 \tag{2}$$

Where F is the surface water inflows from the SRS northern boundary, R is the precipitation, and L is the sum total of water losses from the watershed that result from surface outflows towards the east and south, evapotranspiration, and percolation. The inflow F in our model closely relates to the structural inflow from the S12 and S333 hydrological structures, which is the decision variable that SFWMD regulates. Childers and Leonard (2005) found that the velocity of the freshwater flow varied between seasons and between slough and sawgrass ridges. They estimated the mean velocities of 0.50 cm sec^{-1} and 0.34 cm sec^{-1}, respectively. At these velocities, we expected one to two-month lag between the freshwater inflow at the northern boundary and the coastal freshwater discharges. We estimated the coefficients of the SRS freshwater outflow equation in (2) with different lag periods, but found the one-month lag model to be the best fit.

By plugging (2) into (1), we can directly link the fishery productivity in the SRS coastal area with the managed SRS structural inflows (i.e., combined S12 and S333 structural inflows) along the northern boundary of SRS. That is, we can easily show that

$$C_m = f(F_{m-1}) \tag{3}$$

Creel surveys, taking their name from the wicker baskets anglers use to hold fish, target recreational anglers in a given fishery to estimate total catch and effort. The ENP agents have been interviewing randomly selected recreational anglers over the last 50 years at Flamingo and Chokoloskee/Everglades City boat launch sites upon return from fishing trips on weekends and on some weekdays. Data gathered include the area fished, number of fish kept and released, time expended, and species preference (Osborne et al., 2006). Using this data, we computed CPUE by taking the ratio of the number of fish caught by each angler to effort expended by that angler in hours. Specifically, the CPUE was computed as the total number of fish caught (kept and released) by all anglers in a trip divided by total time expended (hours fished by those anglers). That is,

$$CPUE = \frac{Kept\ +\ Released}{Hours\ fished\ \times\ Number\ of\ anglers\ in\ the\ trip}$$

Finally, *C* for a given species and month was computed by taking the average of species-specific CPUEs of all the anglers surveyed during that month.

The data on hydrological variables in equation (1) and (2) were obtained by running the South Florida Water Management Model (SFWMM) exclusively for the SRS watershed. SFWMM is a physically-based regional-scale simulation model that combines the hydrology and management aspects of water resources from Lake Okeechobee to Florida Bay (South Florida Water Management District, 2005). The model is often referred to as the 2x2, as it has a 2-mile by 2-mile fixed-resolution grid system covering an area of 7,600 square miles. Major components of South Florida’s hydrologic cycle are simulated on a daily continuous mode using climatic data for the 1965-2005 period-of-record. Components include rainfall, evapotranspiration, surface and groundwater flow, seepage, and percolation.

Previous recreational studies (Johnston et al., 2011; Schultz, Johnston, Segerson, & Besedin, 2012), our own consultation with user groups, and our preliminary survey of ENP anglers revealed that recreational anglers do value the overall health of the natural area. But as may be expected, there is no single indicator that fully captures the health or integrity of an entire ecosystem and thus could function as a metric of restoration success. For instance, Ogden et al. (2014) recommended using the abundance of a suite of waterbirds as an indicator of ecosystem health in the coastal marine environment of South Florida, while Harvey et al. (2012) and Mazzotti et al. (2008) concluded that American alligator abundance is "an indicator of ecosystem responses to Everglades restoration because it is sensitive to hydrology, salinity, and system productivity, all factors that are expected to change as a result of restoration." The Science Coordination Team of the South Florida Restoration Task Force established by the U.S. Congress has recommended eleven system-wide ecological indicators in order to understand how the ecosystem is responding to management efforts under the CERP (http://141.232.10.32/pm/recover/perf_ge.aspx). These indicators include abundance of crocodilians, fish and macroinvertebrates, periphyton invasive species, and aquatic vegetation, among others (Brandt et al., 2012; Doren et al., 2009). While there appears to be considerable disagreement among scientists as to which indicator, or group of indicators, best describes the ecosystem responses, there is certainly agreement on the fact that all of these indicators have strong dependencies on hydrological conditions, particularly the extent, duration, and timing of marsh flooding (Holling et al., 1994; Ogden et al., 2005). The dependency is captured by the inundation pattern or hydroperiod of wetlands, as told by marsh depth. For instance, the availability of water during both the wet and dry seasons seems to be the limiting factor for species sustainability and recovery of oysters, spoonbills, pink shrimp, submersed aquatic vegetation, and crocodilians (Brandt et al., 2012). Insufficient water and rapid reversals in water height either during marsh flooding or draining have kept many of the eleven indicators below targets.

For lack of a single comprehensive ecological benefit-relevant indicator, we used the water depth (D_m) as a proxy for the overall ecosystem health. Further, in order to keep the model simple, we considered the above depth-ecohealth relationship only for below-target flow levels, although excess water level could also disrupt wildlife habitat (Brandt et al., 2012). Depth variable data from four observation stations along SRS was averaged using a data set extending from January 2002 to December 2014. Depth was assumed to be the function of surface water inflows through the hydrological structures along the northern SRS boundary (F_m); rainfall (R_m); and the sum total of various losses (L_m) including lateral outflows of the SRS watershed in all directions, evapotranspiration, and percolation. Unlike CPUE (Equation [3]), depth is modeled using seasonal (quarterly) time series variables, thus no lag is assumed. Formally,

$$D_m = a_{31}F_m + a_{32}R_m + a_{33}L_m + \varepsilon_3, \tag{4}$$

where *m* here refers to quarter.

The depth variable in the above equation refers to the level of the water surface with respect to a given gage datum, in this case NAVD 88. The datum is used as a zero point for measurement of water level. The zero point may not correspond exactly to the ground surface elevation at a given location (Holmes Jr. et al., 2001). For example, a location may have an elevation of 4.01 ft above NAVD 88, and a stage of 4.65 ft. Consequently, water depth is calculated as the difference between water level and elevation. Daily median water depth for four stations along Shark Slough (MO-215, NP206, P33, and P34) was averaged and used to calculate mean monthly water depth.

We detected the presence of first-order autocorrelation in the error terms of all the three hydro-ecological models (equations 1, 2, and 4). We resolved the autocorrelation problem by using the Cochrane-Orcutt Procedure (Cochrane and Orcutt, 1949). In all but one case, the serial correlation was removed after the first round of transformation of model variables. Only in the case of equation 4 (the depth-flow model), we had to apply the Cochrane-Orcutt transformation twice.

Penalty function development

The penalty in our study is defined as the periodic loss in the recreation-related ecosystem services suffered by anglers when the freshwater inflows in SRS falls below certain target levels (a management decision or natural shortage of water), or because of changes in natural factors such as rainfall, evapotranspiration, and outflows in the SRS watershed itself. Since the focus of our study is the effect of managing inflows at the SRS northern structures (S12+S333), we construct the penalty function in relation to the flow shortages at those structures in relation to certain target flows. These target flows are derived from the Natural System Model (NSM) (VanZee, 1999), a simulation model that is maintained and run by the South Florida Water Management District (SFWMD) to characterize pre-development hydrologic conditions of the Everglades system. The NSM-based target flows therefore mimic natural hydrologic conditions prior to channelization projects and associated hydrologic alterations in the area in the early 1900s. Later in the paper, we will see that the targets are significantly higher than the average flows since 1990s and even larger than the average flows in much recent years (2012-14).

Let F_m^k be the current monthly SRS inflow at S12+S333 structures, and C_m^k be the current levels of fish catch. Express the flow-induced catch rate $C_m = f(F_{m-1})$ of a species during a given month as percent change from its current level of catch C_m^k as,

$$\Delta C_m = 100\left[\frac{C_m(F_{m-1}) - C_m^k}{C_m^k}\right] \tag{5}$$

Define w_c as the marginal WTP of anglers for a percent change in catch, which will be described later in the discrete choice model. Then

$$\Delta Y_{c,m} = w_c \Delta C_m, \tag{6}$$

where $Y_{c,m}$ is the hypothetical monetary value of the overall recreational fishery catch and $\Delta Y_{c,m}$ is the monetary value of the change in catch rate ΔC_m valued at \$$w_c$ per percent change.

The variable $\Delta Y_{c,m}$ can also be interpreted as the additional price that an average angler would be willing to pay over and above the value that he or she is enjoying at the current catch rate (Y_c^k). That is,

$$\Delta Y_{c,m} = Y_{c,m} - Y_c^k \tag{7}$$

Equating (6) and (7), substituting in (5) for ΔC_m, and simplifying the results, we obtain,

$$Y_{c,m}(F_{m-1}) = Y_c^k - 100 w_c + \frac{100 w_c}{C_m^k} C_m(F_{m-1}) \tag{8}$$

Let $a_m$be the number of anglers' trips in month *m* and $Z_{c,m}$ the total recreational fishery catch value from all trips. Therefore, we express $Z_{c,m}$ as,

$$Z_{c,m}(F_{m-1}) = a_m Y_{c,m}(F_{m-1}) \tag{9}$$

Note that $Z_{c,m}$ is an increasing function of freshwater inflow. We can now formulate the total fishery catch penalty [$P_{c,m}(F_{m-1})$] of not meeting the monthly target flow as,

$$P_{c,m}(F_{m-1}) = Z_{c,m}(F_{m-1}^t) - Z_{c,m}(F_{m-1}), \tag{10}$$

where F_{m-1}^t is the flow target in *m-1*. Figure 3 represents equation (10) where in the amount total penalty decreases as the volume of flow increases, and the penalty reaches zero when the inflow volume reaches the monthly target. We assume zero penalty for $F_{m-1} > F_{m-1}^t$.

By substituting (8) into (9) and the results into (10), we can further simplify fishery catch penalty function as,

$$P_{c,m}(F_{m-1}) = 100 a_m w_c \left[\frac{C_m^t(F_{m-1}^t) - C_m(F_{m-1})}{C_m^k(F_{m-1}^k)}\right] \tag{11}$$

Note that $P_{c,m}(F_{m-1})$ is the difference between catch rates at the target flow (F_{m-1}^t) and the actual flow (F_{m-1}) for a given month, weighted by the catch rate at the current flow (F_{m-1}^k), and multiplied by the value of a percent change in catch (w_c) and the number of total trips (a_m) for the given month. Penalty is lagged by a period because of the lagged catch-flow relationship in (3). Also, the flow-induced shortage in catch in (11), $C_m^t(F_{m-1}^t) - C_m(F_{m-1})$, is above weighted by the current catch rate $C_m^k(F_{m-1}^k)$. Weighting is done because the WTP value in the above equation, w_c, reflects the average angler's willingness to pay for a percent improvement in catch from the current fish catch rate.

While anglers target different species during fishing trips, their preference may vary from species to species. As there are five major species, $i = 1,2, \ldots, 5$, we can obtain the aggregate catch penalty function [$P_{c,m}^a(F_{m-1})$] as a weighted average of individual species catch penalties,

$$P_{c,m}^a(F_{m-1}) = \sum_{i=1}^5 \omega_i \left\{100 a_m w_c \left[\frac{C_{i,m}^t(F_{m-1}^t) - C_{i,m}(F_{m-1})}{C_{i,m}^k(F_{m-1}^k)}\right]\right\}, \tag{12}$$

where ω_i is the weight of the species *i* in terms of anglers' preference given to it during the fishing trip. We require that

$$\sum_{i=1}^5 \omega_i = 1$$

As mentioned before, the water depth in ENP is the key driver of the overall health of the ecosystem. A change in the D_m variable from the target condition is considered as an indication of change in ecosystem health. Recall equation (4) which connects the water depth [$D_m(F_m)$] to water management, i.e., managed flow variable, F_m. We used this equation (4) to link reductions in managed flow from the target level to proportionate changes in the depth variable, and in turn, to proportionate changes in overall ecosystem health using the ratio, $\frac{D_m^t(F_m^t)-D_m(F_m)}{D_m^k(F_m^k)}$. We recognize that this is a simple and broad measure of ecological outcome of a management action. In actuality, indicators of overall ecosystem health may vary from turbidity and seagrass density to presence of particular species of wading birds and alligators and healthy mangroves (Brandt et al., 2012). Further, the above ratio is only a linear and instantaneous representation of ecohealth-flow response while the actual ecosystem response could be non-linear, especially over the long term. Measurement and valuation of more complex ecological functions and service outcomes of management flow are beyond the scope of our study. As the focus of our analysis was the valuation of ecosystem services that were relevant to common users like recreational anglers, it was necessary to keep the measure simple and meaningful to foster better grasp of the measure by the anglers and others. Following Johnston et al. (2012) and Mitchell and Carson (1989), to quantify both intermediate and final ecosystem services, overall ecosystem health was included as a "holistic measure of the ecosystem condition in survey scenarios to quantify this final ecosystem service."

The ecosystem health penalty [$P_{e,m}(F_m)$] is expressed as the dollar value of the percentage change in the depth variable [$D_m(F_m)$], i.e.,

$$P_{e,m}(F_m) = 100 a_m w_e \left[\frac{D_m^t(F_m^t) - D_m(F_m)}{D_m^k(F_m^k)}\right], \tag{13}$$

where w_e is the average angler's willingness to pay in dollars for a percent improvement in the overall ecosystem health (*e*) from the current level.

Combining equations (12) and (13), we compute the total penalty for the fisheries ecosystem services as the sum total of the penalties for lost fish catch and the lost overall ecosystem health due to reduced SRS inflows. That is,

$$P_{T,m}(F_m) = P_{c,m+1}(F_m) + P_{e,m}(F_m) \tag{14}$$

Non-market Valuation of Anglers Recreational Attributes

In order to estimate the anglers' WTP values for changes in recreational fishery attributes, we adapted a discrete choice model (Vojáček and Pecáková 2010), which complies with utility maximization and random utility theory (Lancaster 1966; de Palma 2008). Beginning with a standard random utility specification, an angler is asked to choose among three hypothetical restoration scenarios ($r = N, R_1, R_2$) for ENP ecosystem service restoration. These include a status quo (*N*) option with no restoration and low or no cost and two restoration options (R_1, R_2). Each scenario is characterized by a vector of variables, $Q = [X_1 \ldots X_J]$, representing scenario outcomes. $X_1 \ldots X_{J-1}$ are defined as variables representing ecological outcomes of restoration, *A* represents unavoidable cost, and *S* represents a vector of demographic variables. Following standard notation, that the utility agent derives from option *r* can be represented as

$$U_r(Q, I - A, S) = V_r(Q, I - A, S) + \varepsilon_r \tag{15}$$

where *I* is the disposable income of angler; $V_r(.)$ is a function representing the empirically measurable component of utility; and ε_r is the unobservable stochastic component of utility modeled as econometric error. When presented with a set of scenarios $r = R_1, R_2$, an agent is assumed to choose the one from which he or she derives the greatest expected utility (Train 2009). That is, an agent would say YES to paying an amount A for an environmental improvement if

$$V_1(Q_1, Y - A, S) + \varepsilon_1 \geq V_0(Q_0, Y - A, S) + \varepsilon_0 \tag{16}$$

An agent's WTP is determined by a variety of socioeconomic factors including income, education, and knowledge and use of the resource in question. Thus an important consideration with stated preference is the respondent's information set, which consists of both endogenous factors because of experience or familiarity with the resource and exogenous factors as a result of explicit information presented in the survey instrument (Cameron and Englin 1997; Bergstrom 1990; Freeman 1994). To help ensure agents made informed decisions, a number of multimedia tools were used within the anglers' survey in our study. Two videos, each approximately 1 minute in length, were employed, as were maps of the Everglades and Florida Bay, graphic illustrations, photographs, and text descriptions.

Following the theoretical model, the structure of the discrete choice experiment had respondents choose from three scenarios ($r = N, R_1, R_2$) for restoration of freshwater flow. The questionnaire was developed and tested over one year in a collaborative process that included the participation of economists, ecologists, hydrologists, and members of stakeholder groups, ensuring that relevant attributes were considered (Johnston et al. 2012, Schultz et al. 2012). Respondents were presented with a choice card in which they were asked to select their preferred scenario, valuing percent changes in various fishery attributes and the overall ecological condition from the current level. Johnston et al. (2012) stress the need that a stated preference survey include a comprehensive set of indicators representing both direct and indirect outcomes of management policy that would contribute to respondents' welfare. Failure to do so conveys an 'ambiguous' ecological description of services to the survey respondents. The misrepresentation is characterized as a violation of content validity (Mitchell and Carson, 1989), which could lead respondents to conflate or over speculate the welfare values of those direct indicators (e.g., fish catch, travel distance, etc.) included in the survey (Johnston et al., 2012). To avoid such conflating effect, the choice options in our survey included three attributes characterizing fishing-specific experience (catch rate, size of the largest keeper, and travel distance for fishing) and one attribute representing the overall ecological effect of restoration. We also had the usual price attribute characterizing individual per-trip cost. The combination of distinct fishery-specific and broader ecological indicators will allow respondents to value each of them distinctly. On all choice cards, Scenario I represented the status-quo at low or no additional cost, and Scenarios II and III represented maintaining or improvement of current levels at an increased cost.

Levels for each attribute in the experimental design were assigned using feasible outcomes identified by ecological models and expert consultations. Choice scenarios represented each attribute in relative terms with respect to current conditions, representing a percent change. Table 1 presents different levels chosen for each attribute. A fractional factorial experimental design was used to minimize correlation for a choice model covariance matrix, and the final design consisted of 180 choice profiles blocked into 60 cards (Kuhfeld, 2010; Kuhfeld and Tobias, 2005; Johnston et al., 2013). The survey was implemented using the online Qualtrics platform, and analysis included 600 completed surveys.

The parameters of the random utility discrete choice model in (16) was estimated using the simulated-likelihood mixed logic with Halton draws. As respondents had multiple responses, the model was specified to allow for correlation across their respective responses in the panel data (Johnston et al., 2012). Fixed coefficients were those for catch rate and overall ecosystem health, while size of the largest keeper, travel distance, and additional cost were specified to have random coefficients. Alternative specifications of fixed and random coefficients were attempted before choosing the final model. For instance, we tried a nested logit model as well as models with demographic variables interacting with various attributes. None of those models yielded significant results for the cost parameter. Using the estimated model parameters, we were able to compute the mean WTP of ENP anglers for percent improvements in fish catch (w_c) and overall ecosystem health (w_e). Following standard practice (Hole, 2006; Johnston et al., 2013), the WTP estimates were expressed as the ratios of attribute coefficients to the cost coefficient. Further, the ENP anglers online survey also provided other useful information such as anglers' preference for various species, from which we estimated species weights (ω_i) and used in aggregating the catch-related penalties of model species in equation (12).

Estimation of Monthly Recreational Trips

The penalty function in (9) requires the latest (2015) estimate of the fishing effort in terms of the number of fishing vehicles in the ENP. Osborne et al. (2006) provided historical fishing trip data from 1978 through 2005 in areas 1 to 6 of the ENP, which mostly overlap our study recreational area. During 1978 through 2005 the number of annual recreational vehicles (*A*) ranged from 32,000 (1978) to 38,500 (2005) for the above ENP management areas. In order to estimate 2015 value of *A*, we estimated the following annual vehicle trip model, representing the fishing effort. The variable *A* was assumed as a function of the number of registered recreational vessels in the region (*RRV*) and the U.S. consumer confidence index (*CCI*). Formally, the estimating equation for annual ENP fishing vessel trip was,

$$A = a_{40} + a_{41}RRV + a_{42}CCI + \varepsilon_4 \qquad (17)$$

The *RRV* is an indicator of the overall demand for recreational activities in the region, which we measure using the annual number of recreational vessels registered in Miami Dade, Broward, Palm Beach, Monroe, and Collier counties. These data are available from the Florida Department of Highway Safety and Motor Vehicles (FDHSMV, 2017). The *CCI* variable represents the people's overall financial ability to engage in recreational activities. The University of Michigan (2017) develops this index and makes it available through the Federal Reserve of St. Louis website. We also tried including Florida's population, which was highly correlated with *RRV* and therefore was dropped from the model. The Durbin-Watson test statistic showed that the error term ε_4 was serially correlated. We corrected the model from autocorrelation using the Cochrane-Orcutt Procedure. The estimated model was used to project the annual number of trips for 2015. Total annual fishing trips were further distributed to different months using the seasonal recreational boat distributions estimated by Ault et al. (2008) using an aerial survey of recreational vessels and trailers in ENP waters and parking lots, respectively.

2.3 Results

2.3.1 Shortage in freshwater delivery, depth, and CPUEs.

The current water delivery fell short of the target significantly in the recent years (2012-2014) and the deficit was the highest during the months of March through May (68.3%) and the lowest during the months of September through November (46.1%) (Table 2 and Figure 4). The lowest and highest deficits were found to occur during the months of October and April, respectively. Throughout the study period of 1991 to 2014, actual flow typically came closest to target flow during the wet season, in line with the increased precipitation during those months. The only months in which flow exceeded the target in any year were January 1995, February 1993 and 1995, May 1993, October 1995, and December 1994. The years 1993 and 1995 had the highest levels of flow averaging across all months. The average water depth estimated at the recent average SRS inflows (2012-2014 levels) consistently fell short of the depth to be expected if the freshwater SRS inflow were to be maintained at the target levels. The shortage varied from 82.5% during the months of December through February to 94.5% during the months of September through November.

The estimated catch per unit effort (in fish h^{-1}) were the highest during the summer season (June through August) for all five model species, with 0.37 for snook, 0.29 for redfish, 0.22 for tarpon, 0.77 for snapper, and 0.72 for seatrout. Ault et al. (2008) estimated that the total number of fishing vehicles found in the ENP during the same season was the lowest of all seasons, i.e., only 13.3 percent of the total annual recreational vehicles estimated for the National Park. It was interesting to note that the highest fish productivity was observed when the fishing intensity was the lowest.

However, anglers had suffered deficits in CPUEs for all model species and for all seasons when comparing the model based CPUE at target flow levels to current conditions. The lowest estimated deficits were in the summer months (June through August), probably the result the more than average monthly rainfall during these months compared with the rest of the year in addition to lower fishing intensity. On average, seatrout had experienced the lowest CPUE deficit (27%) while redfish had suffered the highest deficit (41%).

The CPUEs for most study species were fairly constant from 1991 to 2002 across both wet and dry seasons, when snook saw a nearly threefold increase from 2002 until 2009. An extreme cold event in 2010 led to a die-off of snook, with a corresponding increase in CPUE for red drum, possibly caused by decreased predation of juveniles by snook (Boucek and Rehage, 2013; Hallac et al., 2010) or possibly because anglers switched their effort to red drum. By 2013, all species were returning to previous CPUE with a slight upward trend for snapper.

2.3.2 Catch-flow and Stage-flow Relationships

The results of the hydro-biological models are presented in Table 3. All of the model coefficients were statistically significant and had expected signs. The measure of goodness of fit (R^2 value) was higher than 0.4 for all models. The catch-flow model results indicate that surface water discharges from the SRS into the coastal tributaries are the strong determinant of the productivities of the model species. The catch variables were also found to be strongly influenced the seasonal dummy variables. The fall season was used as a trap variable. The catches in all other seasons were significantly higher than the fall season catches. These results are fairly consistent with results from previous studies (Rutherford et al., 1989a, 1989b; Tilmant et al., 1989).

As expected, the SRS freshwater inflow was found to have a positive influence on the average water depth in the downstream watershed. Other variables in the model, rainfall, and all types of losses (i.e., evapotranspiration, percolation, and all lateral outflows combined) also significantly affected the water depth. Finally, the hydrological model, SRS outflow-inflow function, also showed strong results. The effects of SRS inflow and precipitation on SRS discharges were found to be positive, while the relationship between all watershed losses (i.e., evapotranspiration, percolation, and lateral surface water losses) was found to be negative. Again, these results are consistent with the wetland hydrology in general (Dolan et al., 1984) and SRS hydrology in particular (Saha et al, 2014). By combining the results of the last model [equation (4)] with those of catch-flow functions [equation (1)], we can link the fish productivity in the coastal SRS creeks with the SRS northern freshwater inflow, the main management variable of our interest. The integration of the two models will allow us to analyze the effects of changes in freshwater management in SRS on fishery ecosystem system services.

2.3.3 Discrete choice model and annual fishing trips

Table 4 presents the results of the mixed logit random utility discrete choice model of recreational preference. The coefficients of catch and overall ecosystem health were specified as fixed whereas the coefficients of other three attribute variables were specified as random with a normal distribution. We had tried several alternative specifications with different combinations of fixed and random coefficients (Johnston et al., 2012), but chose the one that gave the best results based on statistical significance. All estimated coefficients statistically significant with signs as hypothesized.

As specified in our choice experiment, the coefficients of all attribute variables except the cost variable represent the marginal utility of anglers of increasing or decreasing the attribute levels by a percentage point from their respective reference levels, which in our study reflect the levels for the period when the anglers' survey was conducted, i.e., 2014-2015. The study results indicated that the marginal utility of overall ecosystem health was positive and the greatest of all experiment attributes, followed by the marginal utility of percent change in the size of the keeper or harvest. It is not surprising that sports fishery anglers would care about the size of their keepers (Osborne et al., 2006). The results also showed that the longer the distance that the anglers had to travel for fishing, the less likely that they would choose that plan. That is, anglers suffered disutility with increase in travel distance. Finally, the sign of the coefficient of the cost variable was consistent with our expectation indicating that a restoration plan with increased freshwater was less likely chosen if the costs were higher.

Table 4 also presents the marginal willingness to pay (MWTP) or implicit price of model choice attributes that are associated with increasing freshwater flow in ENP. Marginal willingness to pay can be calculated by taking the ratio of the coefficient of a given attribute variable to the coefficient of the cost variable. As expected, an average angler was willing to pay the highest amount for improving the overall ecosystem health at $3.44 per percent improvement, given all other variables constant. The $3.44 was followed by the MWTP for percent improvement in the size of the keepers ($1.64), a percent reduction in travel distance ($1.58), and a percent improvement in catch ($1.28).

Note that these implicit price estimates of recreational attributes were derived from clearly and unambiguously specified ecological characteristics with quantitative measurements (i.e., in percent changes). The survey had asked anglers if they would pay a given bid amount for a specific (quantitative) percent of improvement in the overall ecosystem health. Therefore, these estimates are likely to be more precise and reliable (Johnston et al., 2012). However, we do recognize the limitation of this method in that anglers were not told what a given percentage improvement in the ecosystem health meant in terms of detailed specifications of system-wide ecosystem indicators (Brand et al., 2012). Anglers were left to make their own subjective judgement of the ecosystem improvement.

The annual fishing trip model which was estimated using the ENP fishing trip data that was available from 1978 to 2005 (Table 3). Both RRV and CCI variables were highly significant determinants of the annual fishing trips. In recent years, both these variables have increased. Using the model parameters and the available estimates of the 2015 registered recreational vessels and reported US confidence index numbers, we estimated the annual 2015 trips at 44,627. This estimate indicated a moderate 16 percent increase in annual trips over the ten-year period beginning in 2005, which saw 38,284 trips. Based on aerial survey data given by Ault et al. (2008) for weekend and weekday samples of fishing boats, we estimated the seasonal distribution of total annual fishing trips to ENP at 17.47% for Fall, 33.04% for Winter, 36.20% for Spring, and 13.29% for Summer. We then equally allocated one-third of each season’s percent of fishing trips to each of the three months of that season. The 2015 estimated annual trip of 44,627 was further allocated to all 12 months of the year. Accordingly, the three Summer months had the lowest number of trips and the three Winter months had the highest number of trips.

2.3.4 Fisheries penalty functions

We used equations (12) to (14) to generate the monthly penalty values with respect to varying levels of freshwater flow at SRS norther boundary through S12 and S333 structures. Table 5 presents the monthly functions. The penalty values are the lost dollar values in recreational experience as a result of shortage in freshwater delivery into SRS in relation to monthly target levels. The penalty reaches zero at the monthly target level. The height of the penalty function varies across the months. During the dry months, November through April, the penalty was found to be high for any given level of flow, whereas during the wet months, May through October, the penalty was found to be smaller than it was in the dry season. Three factors contributed to this variation. During the wet season, the reduction in water shortages in relation to the target delivery reduced the penalty. Also during those months, especially in the Fall, the total number of monthly fishing trips declined. On the contrary, during the rest of the years, either the flow shortage, the number of trips or both were relatively lower than the levels in the dry season.

The slope of the downward sloping penalty curve represents the implicit marginal cost of reducing the water delivery or reallocating the water for upstream uses. The same can be interpreted as the marginal value of increasing the water delivery into ENP in terms of avoided loss in recreational value, i.e., the marginal value of water use for recreation and fishery habitat protection. The monthly recreation marginal value of water ranged from a lowest amount of \$11.88 per acre-feet (AF) to \$112.11 AF^{-1} (Table 5).

Basically, they mirrored the extent of seasonal water shortage and the seasonal recreation demand. The mean annual marginal value (or implicit price) of water was estimated to be \$41.08 AF^{-1}. The major portion of the value can be attributed to the value that anglers attach to overall ecosystem health (\$39.36 AF^{-1}), while a significantly small portion can be attributed to the value anglers attach to fish catch.

The implicit values of water for various uses are not readily available. Frederick et al. (1997) reported water prices in different US economic sectors in 1994 US$. By inflating those values to 2015 using a cumulative inflation rate of 59.9%, we found that their mean estimate of water price for recreation was \$76.77 AF^{-1} in 2015 US dollars. This value was within the range of the monthly water price estimates obtained in this study. Frederick et al. came up with higher values of water for agriculture (\$119.95 AF^{-1}), industry (\$451.00 AF^{-1}), and domestic (\$310.27 AF^{-1}) uses than for recreational uses (\$76.77 AF^{-1}). Our current study was a part of a broad regional research on water resources allocation in South Florida (Mirchi et al., 2018). Two other studies under this broad regional project looked at the value of water for urban and agriculture uses in South Florida. Takatsuka et al. (2018) estimated a much larger value of water at \$280 AF^{-1} for agricultural production, whereas Weisskoff (2018) estimated a marginal price of \$2,000 AF^{-1} for urban uses at about 10% shortfall. South Florida sub-tropical agriculture is known for commercial cash crops such as nurseries, fruit crops, winter vegetables, sugarcane, and citrus. Therefore, one can expect a much higher marginal value of water for use in agriculture than in recreation. Similarly, the fast-growing urban population, real estate, and other businesses tend to push up the value of urban water use.

2.3.5 Simulation of water management scenarios

Table 6 presents the total annual losses in recreational values under alternative water management scenarios. We estimated the total annual penalty values under the baseline and six alternative scenarios. The baseline scenario occurs when the monthly water delivery continues under the current flow rates, which amounted to annual total delivery of 754 KAF. The total penalty was estimated at $25.74 million. The total value is decomposed into two recreational attributes of fish catch at $4.16 million and overall ecosystem health at $21.57 million. We also estimated penalties under six other alternative water delivery scenarios. If the freshwater delivery were to be increased by 50% during all the months (scenario 1), the total annual penalty would be lowered to $22.13 million (a 13.23% reduction in the penalty).

Oftentimes, water management delivery decisions are made for a shorter period of time. Therefore, the next two scenarios considered increase in water flow only a half of the year. Under scenario 2, we increased the flow by 50% only during the dry season, which resulted in the reduction of the losses to $63.37 million, representing 7.92% improvement in avoided losses. Whereas under scenario 3, if we increased water delivery during wet season by 50%, the reduction in recreational losses was much smaller, i.e., penalty was reduced to $65.12 million, representing only 5.37% gain from the baseline penalty. This supports our observations made earlier in the paper that water is more valuable in dry season in terms of providing recreational services.

Two other scenarios 4 and 5 were explored for increasing the monthly freshwater flows to the historical levels (an annual total of 1040 KAF) and by 100 percent of the baseline level (an annual total of 1509 KAF), respectively. While target flow levels are ideal levels to achieve, these two scenarios, along with scenarios 1, 2, and 3, simply reflect incremental policy changes in the quest towards the target flows. Annual flow level of 1991-2005 (simulation 4) is in fact slightly higher than the baseline (2012-14) level (754 KAF) and drastically lower than the target (2,594 KAF). The annual total penalties were reduced to $21.52 million (15.75% improvement) and $18.55 million (26.58% improvement) under scenarios 4 and 5, respectively. By default, if the water delivery were to be restored at the target levels (i.e., to the annual total of 2590 KAF) under scenario 6, the penalty would be completely eliminated. This shows that how far away the current and even the historical water deliveries were from the target, and the respective losses in recreational value were quite substantial on an annual basis. However, we must note that the target levels, determined by the water management agencies, reflect the pre-development water flows. On the other hand, the post-development levels used in the above analysis (scenario 4) refer to the monthly and annual averages for the last 25 years. While the actual flow levels in some of the months during the last 25 year period had reached the respective target levels, restoring the flow to pre-development (target) levels seems unrealistic under the current natural and political environment (i.e., due to the competition from other sectors). The target levels therefore represent at best historic reference levels rather than realistic management goals. For this reason, the comparison of penalties between various management scenarios, all of which have the same reference (i.e., target) levels, makes more meaningful.

2.4 Discussion and Conclusion

An important practical insight became evident from the WTP estimates of various attributes. ENP anglers attached the highest value to improvements in the overall ecosystem health. The case for restoration of freshwater flow in ENP is not just important for improving the fishery habitat (Davis et al., 2005; Chen and Twilley, 1999; Ross et al., 2000). Everglades National Park provides a host of ecosystem services including groundwater recharge, wildlife habitat, carbon sequestration, and mangroves-related services, among others (Richardson et al., 2014; Jerath et al., 2016). Our study clearly shows that recreational anglers do attach highest value on non-fishing related attributes. While the primary focus of anglers during fishing trip may be to catch and harvest as many fish and travel only a reasonable distance to do so, they enjoy other attributes that are indicative of a healthy ecosystem.

As Johnston et al. (2012) note, one of the major limitations of past discrete choice or contingent valuation studies of recreational fisheries is to grossly oversimplify other ecological improvements of a restoration plan (e.g., defining the improvements in low, medium, and high levels). By doing so, the estimates of WTP for fishery improvements could be overestimated as respondents may bundle their value for other ecological aspects of improvements with fishery improvements. Johnston et al. (2012), therefore, used a single composite ecohealth index in addition to fish catch, access, and economic attributes. The WTP for the catch variable turned out be very insignificant upon including the ecohealth indicator variable in their survey. In our study, we used the depth variable as a proxy for ecohealth. Anglers were asked to value percent increase in ecohealth, without being given specific details on the improvements of eleven system-wide ecological indicators (Brandt et al., 2012). Interestingly, with a quantitative value attribution to the overall ecosystem health variable, the WTP value for fishery catch turned out to be small but significant in our analysis. All in all, we find our estimates to be ecologically unambiguous and quantitatively more precise than it would have been without the ecohealth attribute.

The integrated hydro-ecological-economic model developed in our study is probably the first attempt at linking water management variables with Everglades ecosystem services relevant to humans. Although our study considers a single ecosystem service component of ENP, and thus, may seem limited in scope, the approach has potential to assess management decisions in an incremental fashion (Fulford et al., 2016). Past valuation studies on the Everglades ecosystem restoration projects have attempted to measure a larger number of ecosystem services as a bundle of outcomes resulting from large single investment decisions (Richardson et al., 2014; McCormick et al., 2010). While such studies do provide management-relevant information, linking users' preference and behavior explicitly with decision variables yields a powerful management tool. Our model, therefore, has a variety of management applications for water management, not only in ENP, but in other ecosystems dependent on water delivery. The model outcome also lends itself to being an integral component of larger multi-sector optimization models that examine the trade-offs among competing water uses such as environmental restoration; urban use and flood control; and agricultural use. (Mirchi et al., 2018). Further, modeling the avoided losses in economic benefit resulting from incremental increases in freshwater flow allows for evaluation and comparison of restoration scenarios, contributing to benefit-cost analyses.

For instance, SFWMD had considered a number of alternative water delivery plans for South Florida in recent years. In the case of the 2008 Modified Water Deliveries to ENP, Tamiami Trail Modifications, Limited Reevaluation Report (LRR) plan, a 1-mile bridge, other road improvements, and modifications to increase head in the L-29 canal would allow peak freshwater flows into the park at 47% higher rates than current conditions (National Park Service, 2012). The LRR bridge project was completed in 2013. At a 47% increase from the current flow level of 1,848 cubic feet per second (cfs) (National Park Service, 2012) to the project goal level of 4,000 cfs, the penalty value of the recreational fishing experience would be lowered by 13% (derived from the scenario 1 analysis).

One of the significant contributions of this study is to quantify implicit prices of water for recreation and habitat protection. To our knowledge, such information is very scare in the literature. See Frederick et al. (1997) for a most comprehensive list of water prices, which are more than 20 years old. We consider the price estimates in our study to be very conservative since we were able to account for only one major ecosystem value, i.e., anglers' preference for fishing and habitat protection. Other ecosystem service values must be measured and linked to freshwater delivery in order for this price to be complete. However, the price of recreational water use that we developed is comparable to previously available estimate (Frederick et al., 1997).

Our study shows that the total valuation of recreational ecosystem services is sensitive to various ecological, economic, and management factors. The total value of lost recreation benefits is influenced by climatic factors such as rainfall, evapotranspiration, and other hydrological factors. The estimated hydrological equations show statistically significant relationships between these factors and fish productivity. Therefore, future changes in climate could have a significant impact on the valuation of fishery ecosystem services. Biological factors that might affect fish abundance, catch and size of keepers could all significantly affect anglers' preferences, and in turn, the total valuation. Similarly, the future Florida population and anglers' confidence about the economy will have a direct bearing on the future valuation of recreational services.

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Tables

Table 2.1. Attribute Levels in Choice Experiment Design

Variable	**Levels**
Catch rate	• 40% lower than the current level[1] • 20% lower than the current level[1] • 10% lower than the current level[1,2] • Same as the current level[1,2] • 10% higher than the current level[2] • 20% higher than the current level[2] • 40% higher than the current level[2]
Size of the largest keeper	• 20% smaller[1] • 10% smaller[1,2] • Same size as the current largest keeper[1,2] • 10% larger[2] • 20% larger[2]
Boat travel distance for fishing	• 40% increase in the distance[1] • 20% increase in the distance[1] • 10% increase in the distance[1,2] • Same as the current distance[1,2] • 20% decrease in the distance[2] • 40% decrease in the distance[2]
Overall ecosystem health	• 40% worse[1] • 20% worse[1] • Same as the current health[1,2] • 20% better[2] • 40% better[2]
Cost	• $0 cost per trip[1] • $10 cost per trip[1,2] • $20 cost per trip[2] • $30 cost per trip[2] • $40 cost per trip[2] • $50 cost per trip[2]

[1]Scenario 1, [2]Scenarios 2 and 3

Table 2.2. Baseline and target level total regulated freshwater delivery at ENP node (S12 and S333 structures), estimated average depth, and catch per unit effort of model recreational species[1]

Season	SRS Inflows at S12+S333 (KAF)			Estimated Average Water Depth (ft)			Average Catch Per Unit Effort (Per Hour)									
							Snook		Redfish		Tarpon		Snapper		Seatrout	
	Current	Target	Deficit from Target (%)	At Current Flow	At Target Flow	Deficit from Target (%)	Current	Deficit from Target (%)	Current	Deficit from Target (%)	Current	Deficit from Target (%)	Current	Deficit from Target (%)	Current	Deficit from Target (%)
Dec-Feb	258.4	593.3	56.4	0.98	2.55	61.7	0.25	44.4	0.24	38.9	0.16	37.4	0.63	34.3	0.58	30.3
Mar-May	141.9	447.9	68.3	0.26	1.94	86.5	0.24	38.2	0.17	40.2	0.19	28.4	0.55	31.2	0.57	25.2
Jun-Aug	251.2	655.7	61.7	0.77	2.80	72.4	0.23	35.7	0.20	32.8	0.22	22.6	0.62	25.6	0.61	21.1
Sep-Nov	481.7	893.2	46.1	1.22	3.20	61.9	0.16	48.1	0.12	48.1	0.08	48.1	0.25	48.1	0.20	48.1

Baseline levels are based on estimated average historical values.

Table 2.3. Estimated models of catch-flow and depth-flow relationships

Model	Variable	Coefficient	Std Error	Adj R^2	*N*	*Durbin-Watson*
Snook catch [equation (1)]				0.40	179	1.8298
	SRS West Outflow	0.00290*	0.00038			
	Winter	0.14883*	0.02875			
	Spring	0.19960*	0.03094			
	Summer	0.15909*	0.02871			
Red Drum catch [equation (1)]				0.49	179	1.8652
	SRS West Outflow	0.00222*	0.00027			
	Winter	0.16244*	0.02041			
	Spring	0.13772*	0.02193			
	Summer	0.14623*	0.02038			
Tarpon catch [equation (1)]				0.44	179	1.9654
	SRS West Outflow	0.00142*	0.00031			
	Winter	0.11465*	0.02919			
	Spring	0.16529*	0.02787			
	Summer	0.18110*	0.02854			
Gray Snapper catch [equation (1)]				0.58	179	1.9768
	SRS West Outflow	0.00476*	0.00067			
	Winter	0.46159*	0.05544			
	Spring	0.47467*	0.05677			
	Summer	0.49576*	0.05513			
Spotted Seatrout catch [equation (1)]				0.68	179	1.9271
	SRS West Outflow	0.00367*	0.00052			
	Winter	0.45322*	0.04401			
	Spring	0.51085*	0.04456			
	Summer	0.51841*	0.04368			
SRS Outflow [equation (2)]				0.79	489	1.8992
	SRS North Inflow (m-1)	0.36999*	0.01525			
	Rainfall (m-1)	0.11899*	0.01211			
	Evaporation + Percolation + South Outflow (m-1)	-0.10740*	0.01832			
Water Depth [equation (4)]				0.79	59	1.8224
	Intercept	0.65464**	0.12955			
	SRS North Inflow	0.00436*	0.00057			
	Rainfall	0.00142*	0.00036			
	All Losses	-0.00292*	0.00076			
Annual Fishing Trips [equation (17)]				0.49	27	1.5604
	Intercept	8594.26	3040.01			
	Registered recreational vessels	0.09624*	0.03503			
	US consumer confidence	193.24*	63.34			
	2015 estimated # annual trips	44,627				

* $p < 0.01$; ** $p < 0.05$; *** $p < 0.10$

Table 2.4. Mixed logit models of discrete choice experiment and willingness to pay for ENP fishery recreational attributes

Variable	**Coefficient**	**Std. Error**
Catch[a]	0.008138*	0.002580
Ecosystem Health[a]	0.021800**	0.002896
Keeper Size[b]	0.010381**	0.004273
Travel Distance[b]	-0.009992*	0.002653
Cost[b]	-0.006344**	0.003184
Chi-square	17.49	
n	3468	

Attribute	Willingness to Pay	Std. Error
Catch	1.28**	0.67437
Ecosystem Health	3.44**	1.68306
Keeper Size	1.64***	0.93548
Travel Distance	-1.58***	0.85609

[a]Fixed, [b]Random; *$p < 0.01$; ** $p<0.05$, ***$p<0.10$

Table 2.5. Monthly penalty or lost values recreational ecosystem services due to unmet target delivery at S12 and S333 structures along the SRS northern boundary

Freshwater Flow (KAF)	Jan	Feb	Mar	April	May	June	July	Aug	Sept	Oct	Nov	Dec
						(Million $)						
0	2.15	1.51	7.45	6.83	3.26	0.74	1.26	1.87	1.44	1.58	1.81	2.98
50	1.58	0.93	5.50	4.86	1.38	0.47	1.01	1.51	1.14	1.31	1.55	2.45
100	1.02	0.35	3.54	2.89	0	0.20	0.76	1.16	0.84	1.05	1.29	1.91
150	0.46	0	1.59	0.91	0	0	0.51	0.81	0.55	0.78	1.03	1.38
200	0	0	0	0	0	0	0.26	0.45	0.25	0.51	0.77	0.84
250	0	0	0	0	0	0	0.01	0.10	0	0.25	0.52	0.30
300	0	0	0	0	0	0	0	0	0	0	0.26	0
350	0	0	0	0	0	0	0	0	0	0	0	0
Marginal value ($/AF)	28.46	28.73	111.79	112.11	110.26	13.79	13.39	15.46	12.63	12.03	11.88	27.91
Mean marginal value (Min – Max) ($/AF)										**41.54 (11.88 - 112.11)**		
Mean marginal value for ecosystem health only (Min – Max) ($/AF)										**39.36 (10.05 - 109.05)**		
Value of water in the US ($/AF) [Source: Frederick et al., (1997)]:								In 1994 US $			In 2015 US $[a]	
Recreation/habitat								48.00			76.77	
Irrigation								75.00			119.95	
Industrial								282.00			451.00	
Domestic use								194.00			310.27	
Thermal power								34.00			54.38	
Hydropower								25.00			39.98	

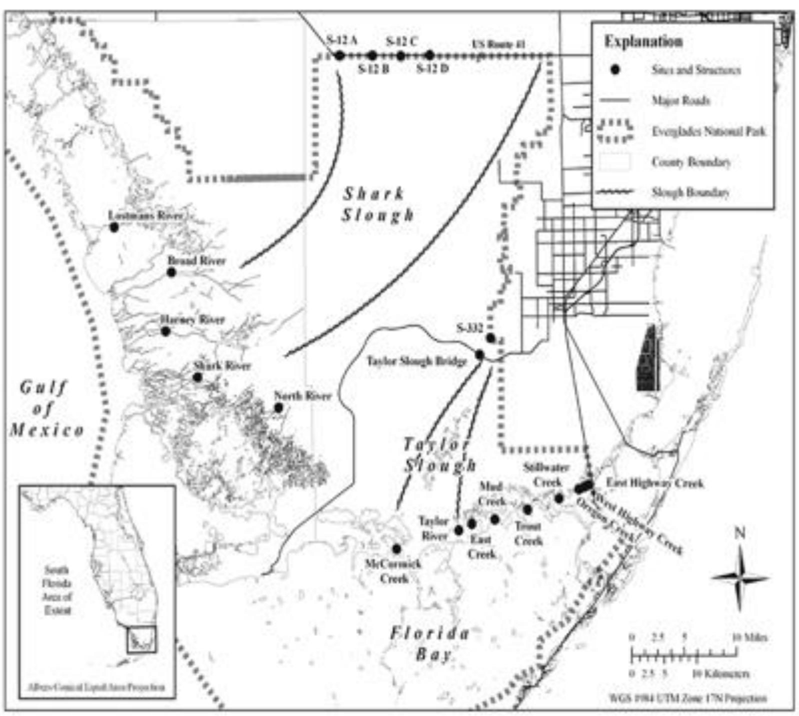
[a]Assumed a cumulative inflation rate of 59.9% between 1994 and 2015.

Table 2.6. Effects of alternative water management on losses in recreational ecosystem service values

Regulated Water Flow Scenarios	Annual Delivery (KAF)	Penalty (Million $)	Gain in Recreational Value from the Baseline (%)
Baseline	754	25.72	0.00
Increase by 50% all months (scenario 1)	1132	22.13	13.96
Increase by 50% dry months (scenario 2)	766	23.70	7.85
Increase by 50% wet months (scenario 3)	1043	24.14	6.14
Increase to historical flow (scenario 4)	1040	21.52	16.33
Increase by 100% all months (scenario 5)	1509	18.55	27.88
Target level delivery (scenario 6)	2590	0.00	100.00
			Percent of Baseline Total
Baseline – ecosystem health only	754	21.57	93.96
Baseline – recreational fishing only	754	4.15	6.04
Baseline	754	25.72	100.00

Figures

Figure 2.1. Map of Everglades National Park: Shark River boundary, the location of S12 and S333 hydrological structures and the southwest outflow tributaries



(Source: https://sofia.usgs.gov/publications/papers/swdis_salmon/images/fig1x.gif)

Figure 2.2. Integrated framework for developing ecological-economic penalty function for managing freshwater flows in the Florida Everglades

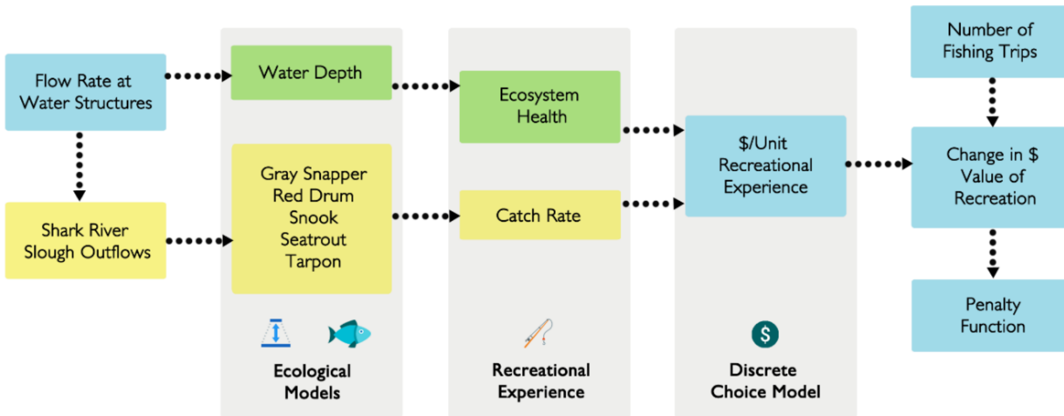


Figure 2.3. Total economic recreational catch value in relation to flow

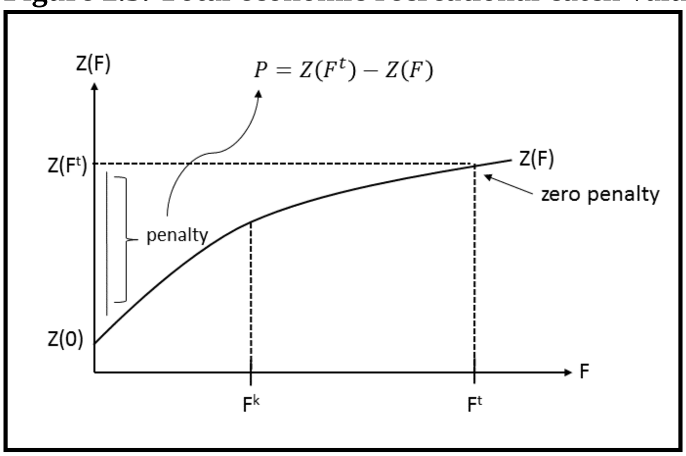
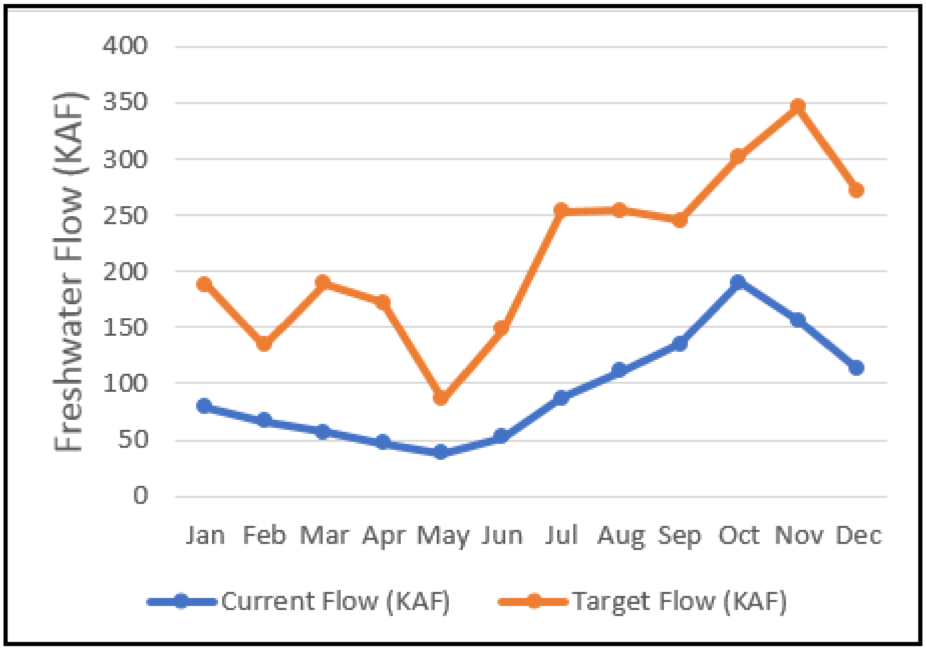


Figure 2.4. Three Year Average Current Flow and Target Flow at the ENP Node (S12+S333 Structures)



3. Valuing Ecosystem Services Under Climate Risk: A Case of Recreation in the Florida Everglades

3.1. Introduction

3.1.1 Effects of sea level rise on the Florida Everglades

The Intergovernmental Panel on Climate Change (IPCC) report outlines changes in the frequency, spatial distribution, and magnitude of several climatic conditions and extreme events that are likely to occur in the not too distant future and could pose significant risks to human well-being (IPCC, 2014). Among such changes is accelerated sea level rise (SLR), posing particular concern to coastal communities. South Florida is among the areas of North America most vulnerable to inundation (Dolan & Walker, 2006; Erwin, 2009; Gornall et al., 2010; Scavia et al., 2002). In addition to inundation, rising sea level can cause shoreline erosion and inland migration, altered salinity and water quality regimes in coastal bays, and increased salinity of freshwater ecosystems and aquifers (Scavia et al., 2002).

Increasing threat of SLR and other adverse environmental phenomena (pollution, etc.) inflict significant losses on society. In the context of SLR and Florida residents, Meng (2016) shows that people are willing to implement and support adaptation plans. However, the extent of their support to adaptation plans depends on, among other factors, their perception of SLR risks, where they live, and whether they are residents or not. Obviously, the more concerned people are about future climate, flood and pollution risks, the more eager they are to support and value an adaptation plan (Hunter et al., 2012; Lee & Cameron, 2008; Meng, 2016; Schaafsma, Brouwer, & Rose, 2012). Studies also show that resource users attach significant non-market benefits to climate adaptation plan, for instance increasing freshwater flows in the Everglades in terms of recreational and commercial fisheries (Brown et al., 2018), groundwater improvements (C. J. Richardson, 2010; L. Richardson et al., 2014), and carbon storage (Jerath et al., 2016).

However, studies that explicitly link the users' perception of climate change and other risks to their welfare value estimates are limited. A few notable exceptions that link environmental risk perception and welfare estimation include Viscusi and Zeckhauser (2006) and Lee and Cameron (2008) in the case of climate risk reduction. Using a choice experiment, Brouwer and Schaafsma (2012) showed that homeowners' willingness to pay for flood insurance depends on where people live (along the coast or the river) and their perceptions about flood risks. Birol et al. (2009) also showed that local residents in their study were found to be willing to accept an increase in local taxation to reduce flood risks. Remoundou et al. (2015) found that in Northern Spain, people concerned about SLR and ocean temperature were willing to pay higher values to protect biodiversity and recreation opportunities.

On the contrary, a recent study in *Science* by Stern et al. (2016) notes the presence of a certain degree of neoskepticism about climate risks, generally accepting the existence of anthropogenic climate change but "advocat[ing] against urgent mitigation efforts," which can be a major challenge for resource managers to gain public support for mitigation (Moser & Ekstrom, 2010). Neoskeptics who defend business as usual may not necessarily disbelieve in climate change, but may want to see more scientific evidence than what is available, think it is not happening during their lifetimes, or defer responsibility to the government or the future. Understandably, neoskeptics may not readily support public or private actions for mitigation and see such as an economic burden.

The purpose of the current paper is to first develop estimates of various psychometric risk measures that characterize people's risk perception (RP), risk concern (RC), and risk-reduction action (RR), and then to test how their risk perception affects their willingness to pay for risk reduction. The case in point is recreational fisheries in the Florida Everglades and the non-market benefit that anglers derive from increasing freshwater flow as a measure to mitigate the effects of SLR on the coastal ecosystem. The study also aims to assess if there is an element of risk skepticism present among the recreational resource users of the study ecosystem. The present study follows Hunter et al. (2012) in modeling psychometric measures of RP, RC and RR and then incorporates those measures into conventional utility-theoretic model of non-market valuation. The study makes two noteworthy contributions to the climate-related risk valuation literature and its management. First, current research on the effects of climate risk perception on adaptation and valuation is rather limited particularly in the context of SLR and coastal resource protection. The study sheds light on how different psychological phases of risk evolution—RP, RC, and RA—influences the environmental value construct (willingness to pay). Second, although certain stakeholders may be apathetic to climate risks, the impacts those risks will have on society are real. Understanding the demographic and other differences in people's risk perception and valuation could aid resource management agencies in targeting stakeholders and designing programs that would improve climate risk literacy and awareness, and in turn, their support for climate risk mitigation (Vignola, Klinsky, Tam, & McDaniels, 2013).

3.1.2 Estimating the benefits of improved recreational fisheries

Lancaster's theory of value provides the conceptual microeconomic framework for choice modeling (Hanley, Mourato, & Wright, 2001; Hoyos, 2010), which centers upon the assumption that an agent's utility or net benefit received from a good can be decomposed into utilities for the good's composing characteristics (Lancaster, 1966). That is, an individual received utility not from the good itself, but from the characteristics or attributes of said good.

The discrete choice experiment (DCE) methodology elicits individual's preferences through constructing a hypothetical market scenario using a questionnaire. Respondents are presented several choice sets consisting of mutually exclusive alternative descriptions of a good from which they select their most preferred alternative (Hanley et al., 2001). Consumer decisions can be separated into a discrete choice, i.e., which good to consume, and a continuous choice, i.e., how much of that good to consume. Discrete choice experiments are constructed to isolate the discrete choice, making the methodology ideal for valuation of non-market goods such as ecosystem services in that the quantity of these goods are fixed for all agents (Hanemann, 1984). By their very nature DCEs force individuals to weigh the trade-offs of present costs or benefits, which are known with certainty, against risky future outcomes (de Palma et al., 2008). When the price of a good is included as an attribute, willingness to pay (WTP) for changes in attribute levels can be recovered (Hoyos, 2010).

The authors adapted a discrete choice model (Vojáček & Pecáková, 2010) complying with utility maximization and random utility theory (de Palma et al., 2008; Lancaster, 1966) to estimate resource users' WTP for improved recreational fishery ecosystem services. Following a standard random utility specification, a resource user is asked to select one of three hypothetical scenarios, two of which are restoration options at a higher cost than the low or no cost status-quo option. Outcomes of each scenario are characterized by a vector of variables (Q), and the utility a resource user derives from option *r* can then be represented as

$$U_r(Q, I - A, S) = V_r(Q, I - A, S) + \varepsilon_r \tag{1}$$

where *I* is the resource user's disposable income, *A* is the unavoidable cost the resource user would be willing to pay for the improved environmental quality, and *S* is a vector of demographic variables. The observable, or empirically measurable, component of utility is represented by $V_r(\cdot)$, while the unobservable stochastic component is represented by ε_r and modeled as econometric error. A resource user is assumed to choose the scenario from which they derive the greatest utility (de Palma et al., 2008). That is, they would be willing to pay an amount *A* if,

$$V_1(Q_1, I - A, S) + \varepsilon_1 \geq V_0(Q_0, I - A, S) + \varepsilon_0 \tag{2}$$

While socioeconomic factors including income and education are determinants of an individual's WTP, it is also influenced by their perceptions of and preferences for risk (Bartczak, Mariel, Chilton, & Meyerhoff, 2013; O'Connor, Bord, & Fisher, 1999). Following Johnston et. al (2013), alternative-specific constants indicating improved environmental quality scenarios are used in model estimation.

Additionally, a resource user's knowledge of the resource, both endogenous and exogenous, also has a bearing on their WTP, thus any respondents who had not fished in the Everglades within the previous three years were screened out of the survey, guaranteeing at least a minimum level of experience and endogenous familiarity with the resource. To ensure respondents were making informed choices, additional information about the Everglades area was presented in the survey (Bergstrom, 1990; Cameron & Englin, 1997; Freeman, 2003), including maps, graphic illustrations, text descriptions, and videos.

Simulated-likelihood mixed logit was used to estimate the parameters of the random utility discrete choice model. Using these estimated model parameters, the authors calculated the mean WTP of ENP anglers for percent improvements in fish catch (w_c) and overall ecosystem health (w_e), expressed as the ratios of the attribute coefficients to the cost coefficient (Hole, 2006; Johnston et al., 2013).

3.1.3 The effect of risk perception on WTP

Individuals may value ES for a range of reasons. Pure non-use value, or existence value, is an individual's WTP for simply knowing that a resource exists even if no use is indented. Utilitarian, or use value, is the usage of an ecosystem for amenities or products that derive both present and future benefits. For example, consumptive products such as timber and amenities such as recreation are considered use values (Costanza, Farber, & Maxwell, 1989). Where existence and use values have more than a modicum of certainty attached, option value is a function of uncertainty (Pearce & Turner, 1990). Because individuals are uncertain about future demand, one may be willing to pay now for the option of using a resource in the future (Hein, van Koppen, de Groot, & van Ierland, 2006). The option value is a premium that an individual will pay above the expected use value, resulting from uncertainty about either an individual's preferences or the price or availability of the resource in the future, and is conditional upon the individual being at least to some degree risk averse (Costanza et al., 1989; Hein et al., 2006; Pearce & Turner, 1990).

Individual risk perceptions may be influenced by the context in which individuals make a decision (James & Meek, 1976; Slovic, 1987). Context can correspond to past experience, anticipatory feelings regarding some future state, or even to the way in which decision outcomes are presented (Cohen, Etner, & Jeleva, 2008; Tversky & Kahneman, 1986). Stone and Gronhaug (1993) classified the components of perceived risk as: financial, psychological, social, performance, physical, and time-related. Berk and Fovell's (1999) study on public perceptions of climate change presented a sample of Los Angeles residents with sets of empirically possible and historically plausible hypothetical climate scenarios to estimate their WTP to prevent significant climate change. Among the findings were that respondents were more concerned with use value than existence value, and that it would take very large climate changes from a scientific perspective to generate even modest changes in WTP. Respondents' perceptions of climate change were found to be relative, as evidenced by valuation of future climate change being a function of the respondent's current climate. Ultimately, they found that within the range of climate scenarios considered, individuals would be willing to incur at least some costs to prevent change (Berk & Fovell, 1999).

An understanding of public perceptions of climate change risk is critical in order to avoid social barriers and garner public support for mitigation and adaptation policy implementation (Vignola et al., 2013). In the context of health risk, Mitchell (1998) finds that beyond a certain threshold or tolerance level of perceived risk, individuals will employ risk-reduction strategies to lower the perceived risk to or below the tolerance level. The argument that a higher perceived risk leads to a higher motivation to adapt to climate change is also supported by Osberghaus et al.'s (2010) study on information and risk perception, which builds on the socio-psychological model of Protection Motivation Theory. The approach the authors chose in the present study to examine perceived risk is the psychometric paradigm, wherein individuals make quantitative decisions regarding the current and desired risk levels of certain hazards, their desired level of regulation of each (Fischhoff, Slovic, Lichtenstein, Read, & Combs, 1978; Slovic, 1987; Slovic, Fischhoff, & Lichtenstein, 1984), and express their willingness to pay for risk reduction (Hunter et al., 2012; Sukharomana and Supalla, 1998; Georgiou et al., 1998). Following Fischhoff et al. (1978) and others, the authors assume that those who perceive climate-related risks to be real and high, become more concerned about its adverse impacts on their future availability of certain ecosystem services that they enjoy currently. Concerned users will further show higher willingness to take certain actions to mitigate the risks. Such actions may be to push for more regulation and public investments for risk mitigation and willingness to make their own contributions for ecosystem improvements, expressed as increased welfare values (see Figure 3.1).

3.2. Study area

The largest subtropical wetland ecosystem in North America, ENP comprises approximately 1.5 million acres on the southern tip of the Florida peninsula, including Florida Bay. Containing both subtropical upland and marine ecosystems, freshwater slough and seasonally flooded marl prairie, tropical hardwood hammock forest, pine rockland, and mangrove and estuarine habitats (Gunderson, 1994; C. J. Richardson, 2010; Saha et al., 2012), ENP's aquatic communities support a variety of seagrasses, freshwater benthic plants, and aquatic organisms. Not only does the Everglades provide vital breeding and foraging habitat for over 400 species of birds, functioning as an important migratory corridor, it is also critical for water storage and recharge of the Biscayne aquifer, the principal source of freshwater for south Florida (Lorenz, 2014; Saha et al., 2012).

Everglades National Park and its surrounding bays, including Whitewater Bay, Tarpon Bay, and Florida Bay, generate more than $1.2 billion in annual economic activity from its world-class recreational fisheries (Fedler, 2009). Recent annual fishing reports estimate that over 90% of boaters in ENP participate in recreational fishing, with boaters launching primarily from Flamingo and Chokoloskee (Ault et al., 2008). Brown et. al. (2018) estimated the number of unique fishing trips in ENP at 44,627 in 2015, a 16% increase over the preceding ten-year period. This estimate is a function of the number of registered vessels and consumer confidence, both of which have been increasing in recent years. Based on aerial surveys, the seasonal distribution of fishing trips peaks in winter (33.04%) and spring (36.2%), falling in summer (13.29%) and fall (17.47%) (Ault et al., 2008).

3.3. Survey design

3.3.1 Survey instrument

Data for this study were gathered from responses to an online questionnaire administered in November and December 2015. The questionnaire was developed and tested over one year in a collaborative process, including the participation of ecologists, hydrologists, economists, and stakeholders, ensuring that relevant attributes were considered (Johnston, Schultz, Segerson, Besedin, & Ramachandran, 2012; Schultz, Johnston, Segerson, & Besedin, 2012). Screener questions were employed to ensure respondents were at either full- or part-time residents of the state of Florida and had visited ENP for recreational fishing at least once in the previous three years. A total of 3,354 questionnaires were attempted, of which 2,949 (87.92%) were completed. Of those, 600 (20.34%) were usable, with the remainder discarded because of nonresponse to specific questions (e.g., choice experiment).

The questionnaire was structured into three sections. The first section explained the motivation and purpose of the survey, along with an assessment of respondents' knowledge and use of the Florida Everglades, including questions about how often they fish in the Everglades, where they fish, and what species they target. The second section provides a non-technical explanation of climate change, SLR, and the possible effects of both on the Everglades ecosystems and the fish populations. This explanation sets the context for the discrete choice experiment, also in the second section, with respondents each receiving two randomized choice cards. The third section explores respondents' perceptions of and concern about climate change and SLR, both in general and regarding the Everglades in particular, along with their attitudes toward control or regulation of these risks. These eleven items, following the psychometric paradigm (Slovic, 1987), were rated on a five-point Likert scale. The first set of three questions evaluated respondents' perception of the risk of SLR [Risk Perception (RP) variables]. The second and third sets of four questions each evaluated respondents' concern [Risk Concern (RC) variables] about specific risks and attitudes toward control or regulation [Risk Reduction or Regulation (RR) variables], respectively. 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 0.4, 0.9, 0.6 and 0.8, respectively. The authors recognize that the RP questions may have failed the internal consistency test with alpha = 0.4 < 0.6. However, other groups are found to be consistent and it is not believed the above issue has affected the Factor Analysis results to be discussed later. The final section also included a series of demographic questions (e.g., age, education, gender, income).

3.3.2 Valuation scenarios

Following the theoretical model, the structure of the discrete choice experiment had respondents choose from three scenarios for restoration of freshwater flow. See Brown et al. (2018) for detailed description of the scenario designs used in the experiment. The survey presented respondents with two sequential randomized choice experiments, each consisting of one choice card in which they were asked to select their preferred of three scenarios in terms of percent change from the current level in five attributes of the recreational fishing experience. Three of these attributes were fishery-specific (catch rate, size of the largest keeper, and travel distance for fishing) and one attribute represented the overall ecological effect of restoration in order to provide a comprehensive array of direct and indirect indicators of management outcomes (Johnston et al., 2012). The final attribute was price, characterizing the individual per-trip cost. All choice cards included a “status-quo” Scenario I, representing no changes to water management at low or no additional cost, with Scenarios II and III representing conditions at least as good or better than the current level at a higher cost (Brown et al., 2018).

Attribute levels were assigned after expert consultation to determine feasible outcomes. Each attribute is represented as a percent change relative to current conditions.

Attribute levels for catch rate were: no change, 10%, 20%, and 40% higher and lower. Levels for size of the largest keeper were: no change, 10%, and 20% bigger or smaller. Levels for boat travel distance for fishing were: no change, 10%, 20%, and 40% decrease or increase. Levels for overall ecosystem health were: no change, 20%, and 40% better or worse. The additional per-trip cost ranged from $0 to $50 in increments of $10. To minimize correlation for a choice model covariance matrix, a fractional factorial design was used, resulting in 180 unique choice profiles blocked into 60 choice cards. The survey was conducted online using the Qualtrics platform, and 600 completed surveys are analyzed in this study.

3.3.3 Econometric model specification

Following previous studies (Berk & Fovell, 1999; Hanley et al., 2001; Hoyos, 2010; Hunter et al., 2012; Johnston et al., 2012; Vojáček & Pecáková, 2010), in the analysis of the responses the random utility models were estimated using simulated-likelihood mixed logit with Halton draws in preference-space, allowing us to consider the coefficients as independent and randomly distributed for all the attributes except Cost. Following the random utility model described in equation (2), a respondent's probability of saying yes to paying amount A is

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

(3)

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

(4)

$$= Fn(n)$$

(5)

$$= Fn(\Delta V)$$

(6)

Where $n = \varepsilon_0 - \varepsilon_1$ and $\Delta V = V_1(Q_1, Y - A, S) - V_0(Q_0, Y - A, S)$, ΔV is the difference in utility, and $Fn(\Delta V)$ is the cumulative probability density function. Per the logit model,

$$Fn(\Delta V) = \frac{1}{1+e^{-\Delta V}} \tag{7}$$

$$Fn(\Delta V(A)) = \frac{1}{1+e^{-\Delta V(A)}} \tag{8}$$

The observable component of utility V_r for each individual *i* is specified to be linear in parameters, such that

$$U_{ri} = \sum_k \beta_{rik} X_{rik} + \varepsilon_{ri} \tag{9}$$

where X_{rk} is a vector of *K* choice-related characteristics consisting of individual characteristics and observed attributes, and β_{rk} is a vector of *K* parameters to be estimated.

In the present study, respondents make a choice among three alternatives: one status quo, and two with some level of restored freshwater flow and improved services compared to the status quo. The restoration can be realized at a cost to be paid in increased boat launch fee per fishing trip, and the cost of no restoration is negligible or zero. Using this, equation (9) can be generally formulated as,

$$U_{ri} = \alpha + \beta_f F_{ri} + \beta_c C_{ri} + \beta_s S_{ri} + \varepsilon_{ri} \tag{10}$$

where α is the alternative specific constant (ASC), β_f is the vector of coefficients to recreational fishery choice attributes *F*, β_c is the coefficient to cost attribute *C*, and β_s is the vector of coefficients to sociodemographic factors and risk attitudinal attributes *S*.

Five specifications of this model were estimated to explore the effects of individuals' risk perceptions on WTP in a step-wise fashion. Model 1 is specified as an "attribute-only" model to set a baseline for testing interaction effects,

$$U_{ri} = \alpha + \beta_f F_{ri} + \beta_c C_{ri} + \varepsilon_{ri} \tag{11}$$

Models 2 and 3 then interact the ASC with a vector of risk attitudes and sociodemographic factors, respectively,

$$U_{ri} = \alpha + \beta_f F_{ri} + \beta_c C_{ri} + \beta_{Aper} A_{ri} Per_i + \beta_{Acon} A_{ri} Con_i + \beta_{Ared} A_{ri} Red_i + \varepsilon_{ri}$$

(12)

$$U_{ri} = \alpha + \beta_f F_{ri} + \beta_c C_{ri} + \beta_{Aage} A_{ri} Age_i + \beta_{Aedu} A_{ri} Edu_i + \beta_{Ainc} A_{ri} Inc_i + \varepsilon_{ri}$$

(13)

where *A* is ASC, β_{Aper} is the vector of coefficients to the interaction of ASC and Risk Perception attributes, β_{Acon} is the vector of coefficients to the interaction of ASC and Risk Concern attributes, β_{Ared} is the vector of coefficients to the interaction of ASC and Risk Reduction attributes, β_{Aage} is the coefficient to the interaction of ASC and Age, β_{Aedu} is the coefficient to the interaction of ASC and Education, and β_{Ainc} is the coefficient to the interaction of ASC and Income.

Models 4 and 5 interact individual Risk Perception factors with fishery attributes to test the influence of these factors on WTP versus an attribute-only model described in equation (11).

$$U_{ri} = \alpha + \beta_c C_{ri} + \beta_{RPcatch} RP_i Catch_{ri} + \beta_{RPkeep} RP_i Keep_{ri} +$$

$$\beta_{RPdist} RP_i Dist_{ri} + \beta_{RPeco} RP_i Eco_{ri} + \varepsilon_{ri}$$

(14)

$$U_{ri} = \alpha + \beta_f F_{ri} + \beta_c C_{ri} + \beta_{RPcatch} RP_i Catch_{ri} + \beta_{RPkeep} RP_i Keep_{ri} + \beta_{RPdist} RP_i Dist_{ri} + \beta_{RPeco} RP_i Eco_{ri} + \varepsilon_{ri}$$

(15)

where $\beta_{RPcatch}$ is the coefficient to the interaction of Risk Perception and the Catch attribute, β_{RPkeep} is the coefficient to the interaction of Risk Perception and the Size of the Largest Keeper attribute, β_{RPdist} is the coefficient to the interaction of Risk Perception and the Travel Distance for Fishing attribute, and β_{RPeco} is the coefficient to the interaction of Risk Perception and the Overall Ecosystem Health attribute. Each of the above models was estimated using mixed logit procedure in STATA software.

3.4. Results

3.4.1 Descriptive statistics

Sample demographics are shown in Table 3.1. The average age of respondents was 38.31 with a median of 35, somewhat lower than Florida's population median age of 41.8. Median household income was very similar to the wider population. Respondents' distribution of race and ethnicity was also very similar to the Florida population, although those identifying as Hispanic were slightly underrepresented in the sample.

Most respondents launched their recreational fishing vessels from either Everglades City (45.02%) or Flamingo (41.24%), with the remainder launching from the Florida Keys (13.75%). The majority of respondents reported fishing in both Everglades backcountry and bay waters (65.88%), with 19.83% restricting their fishing to the bays (brackish or saltwater), and 14.29% to the backcountry (primarily freshwater). The top three fish species targeted by anglers (indicating they frequently or always target them) were Gray Snapper (*Lutjanus griseus*), Red Drum (*Sciaenops ocellatus*), and Snook

(*Centropomus undecimalis*). 56% of respondents considered “being able to see other wildlife” very or extremely important. Sixty percent reported being “very much concerned” regarding the Everglades overall ecosystem health.

3.4.2 Respondents’ risk perception, concern, and reduction

Results of the array of psychometric questions are presented in Table 3.2a. Respondents indicated they were most concerned about the overall ecosystem health and future fish abundance in the Everglades, with mean scores of 4.38 and 4.23, respectively. Results indicate that while respondents believe that the predicted SLR will happen in their lifetime and they have a moderately high level of concern about SLR in the Everglades, they do not believe that the relevant public agencies will manage freshwater flow effectively without their contribution to the effort. Overall, results suggest high support for regulatory action (average score of 4.12) as well as a strong sense of personal responsibility for contributing toward actions to minimize impacts of SLR (4.07).

It is interesting to note that survey respondents scored an average value of 2.77 or lower on all the questions that conveyed a sense of skepticism toward SLR-related risks, i.e., “I can live with the negative impacts of SLR (statement #2; mean = 2.35), “it is too early to worry about SLR…” (statement 3; mean = 2.32), and “the relevant public agencies will manage the freshwater flow (statement #10, mean = 2.77). These results indicated that, on average, sample respondents disagreed with the above statements, signifying a lower level of skepticism about climate risk. That is, they feel they would not want to live the negative impacts of SLR, that it is time to worry about SLR, and that they do not want to simply shun mitigation responsibilities to relevant public agencies.

Table 3.2b presents the differences in anglers' risk attitude and perception across different demographic groups. Earlier studies of general Florida population about climate risks and ecosystem restoration (Sikder, 2016) indicate that there could be significant variation among residents of Florida living at different distances from the coast. That study also found some minor differences across people of different demographic groups. The results of this study show that anglers fishing in the Everglades have uniform risk perception and risk mitigation attitudes across wide demographic groups. The only exceptions were as a function of gender and income regarding RC questions, and across marital status, education, and income with respect to RP questions.

Further, responses to psychometric questions were subjected to principal component factor analysis. Kaiser-Meyer-Olkin values indicated that all variables were suitable for inclusion, and a varimax (orthogonal) rotation was performed. Two meaningful factors with eigenvalues > 1 were extracted and loading on a given factor was assumed if loading > 0.5. Each factor represents a certain underlying (latent) attitude towards climate risks. The statements or variables associated with these factors were labeled "Positive Risk Perception (PRP)" and "Risk Skepticism (RSK) about risk posed by SLR" (Stern et al., 2016) for factors 1 and 2, respectively. Measured (observed) risk variables used in the factor analysis and their corresponding loadings are presented in Table 3.3. These extracted factors closely matched the three sets of psychometric questions presented to respondents in the survey. That is, Factor 1 carried those risk questions that were worded as having concern for the changing environment (e.g., concerns about SLR and its impacts on the Everglades, fish abundance and freshwater), and showed a positive willingness to act for addressing those changes (e.g., both personal support and desire for public support of restoration). On the contrary, all three measured variables that were loaded into Factor 2 reflected a certain degree of skepticism in that respondents did not believe the said environmental changes would be real, could live with the negative impacts, and/or did trust the government would take care of the problem. Although the survey was set up with three sets of risk questions (RP, RC and RR), they were loaded into only two factors, representing the two underlying latent variables, Positive Risk Perception and Risk Skepticism. All the four RC variables, one RP variable (i.e., the perception that SLR will happen during one's life) and two RR variables (i.e., that "government must take action" and that "I am also responsible to take action") had large, positive loadings (>0.5206) on positive risk perception factor (PRP). That is, PRP factor describes variations in those seven variables (Table 3.3) adequately. Similarly, three negatively meaning RSK variables ("I can live with negative impacts…", "it is too early to worry…", and "government will take action…") had large, positive loadings (>0.6972). So, the RSK factor described variation in the three measured statements that signify lack of belief in climate risk, apathy and shunning of responsibility to the government.

To support the paradigm described in Figure 3.1, the coefficients of correlation between the pairs of the three sets of psychometric questions were estimated. That is, the average of scores of all the variables within each of the risk categories were computed first, i.e., RP, RC and RR corresponding to each of the first three boxes in Figure 3.1, respectively. The average scores are described in Table 3.2a and the results of the factor analysis in Table 3.3.

The authors then estimated the Pearson's correlation coefficient between RP and RC, and the correlation between RC and RR. Both these correlation coefficients (0.463 and 0.631, respectively) were positive and statistically significant at a 0.01 level. These correlation coefficients support the authors' hypothesis that the respondents who possess strong (high) perception about SLR and the possible impacts on the ecosystem will be concerned about the said risk and as a result, might provide strong support for appropriate mitigation action and express high willingness to pay for such action. Furthermore, authors found positive and statistically significant correlation between RP and RR variables. That is, people who perceive climate risk to be real and happening in their lifetime were more likely to support risk reduction policies.

3.4.3 Respondents' WTP and the influence of risk perception

Five different random utility choice models were estimated, the results of which are presented in Table 3.4, with descriptions of variables presented in Table 3.5. Because discrete choice models require alternative-specific variables, case-specific variables are interacted. Model 1 simply examined the effects of attributes on the choice outcome for all respondents. Model 2 is an extended model in order to test whether respondents were likely to respond differently to three climate risk perception, concern and reduction variables under business as usual scenario vs. the two improve scenarios. Model 3 is an extended model accounting for socio-demographic heterogeneity. Model 4 is an interaction model to compare the interaction risk terms against Model 1. Model 5 is an extension of Model 4, examining the interaction terms as well as the original attributes.

In the basic attribute only model, ASC was positive and significant, indicating that respondents had some preference toward choosing an improve scenario (moving away from status quo). The coefficients for the choice attributes of Catch and EcoHealth were significant with expected signs, and Cost had a negative sign as expected, implying that a scenario is less likely to be chosen if the cost is higher (Brown et al., 2018). The Keeper and Distance attributes were found to be insignificant. In the basic model, the risk perception variable is implicit and subsumed in ASC. In the model that introduces socio-demographic variables interacted with ASC, Catch, EcoHealth, and Cost are significant with expected signs, while the only significant socio-demographic variable is Education, with a positive coefficient.

Models that introduce risk perception, concern, and reduction (terms interacted with attributes) had a positive and significant ASC, again indicating that respondents had some preferences toward choosing improve scenarios. As hypothesized the coefficient of the cost variable was negative and statistically significant across all the five models. The catch and overall ecosystem health variable coefficients were also positive and significant in all but one model as expected. Overall, Chi-Square values of all the estimated models suggested that they fit the data well.

Willingness to pay (WTP) of respondents can be computed using the following formula:

$$WTP = -\frac{\hat{\beta}_{Attribute}}{\hat{\beta}_{Cost}} \tag{16}$$

Where $\hat{\beta}_{Attribute}$ is the model-estimated coefficient for the attribute parameter in question, and $\hat{\beta}_{Cost}$ is the model-estimated coefficient of the cost parameter.

Table 3.6 presents the estimates of WTP of respondents for two ecosystem services (fish catch and overall ecosystem health) under different levels of risk perception. For comparison, estimated coefficients of attribute-only model (model 1) and coefficients of model with risk-perception (model 5) were used to compute WTP estimates. Individual-specific perceptions of the risks posed by SLR were positively related to WTP, indicating that those who viewed it as a higher risk were willing to pay more. For every one percent improvement in fish catch due to increased freshwater flow (a climate mitigation strategy), WTP ranged from \$0.25 at the risk perception level of 3 (measured on Likert scale) to \$1.00 at the risk perception level of 5. Similarly, the WTP value for a percent improvement in the overall ecosystem health increased from \$0.32 at the risk perception score of 3 to \$2.07 at the risk perception of 5. Authors ignored the model-estimated WTP values for risk perception at Likert scale of 1 and 2 as they were negative. Finally, RP-adjusted weighted WTP was calculated using the sample average percent of respondents that expressed different levels of agreement to risk attitude questions as weights. On average, 4.29% strongly disagreed (Likert scale = 1), 7.51% somewhat disagreed (2), 19.27% neutral (3), 31.88% agreed (4), and 37.04% strongly agreed (5) to the eleven risk questions. The risk-adjusted WTP values for percent improvement in catch and ecosystem health were \$0.62 and \$1.21, respectively.

3.5. Discussion and Management Implications

3.5.1 Nonmarket benefits of increasing managed freshwater inflows

Historically, water flowed south from Lake Okeechobee through the Everglades in a broad, slow-moving shallow river. Since development of a comprehensive freshwater drainage system in South Florida beginning in the early 1900s, these flows have been constrained by a dike and levee system and by urban and agricultural water demands, occupying less than half of their original areal extent and relegating the Everglades to part of a complex watershed management system (Ogden et al., 2005; Sklar et al., 2001, 2005). Because the Everglades watershed is managed primarily for agriculture, flood control, and consumptive uses (Sklar et al., 2005), the flow of freshwater through the Everglades has been reduced, channelized, and otherwise modified, resulting in dramatic changes to biota, salinity regimes, and a variety of ecosystem services in the coastal Everglades (Perry, 2008; Rand & Bachman, 2008).

Large, subtropical bays and estuaries within ENP provide critical habitat for a variety of species, including seagrasses and coastal mangrove communities (Bachman & Rand, 2008). They serve as nurseries for larvae and juveniles of many critical species, including highly sought-after sport fish and wading birds (Lorenz, 2014). Freshwater flows are a key determinant of habitat and fisheries resource productivity (Rudnick, Ortner, Browder, & Davis, 2005; Stabenau, Engel, Sadle, & Pearlstine, 2011; Walters, Gunderson, & Holling, 1992), making the recreational fishing industry in the area a direct beneficiary of improved and sustained fishery habitat (e.g., Boucek & Rehage, 2013).

3.5.2 The importance of risk perception

Overall, the study results provide evidence for the fact that the anglers perceive a high level of climate-related to risks and their impact on the Everglades ecosystem services. More than 68% of the respondents believe risk is real, support mitigation actions and express a high degree of willingness to pay for mitigation. As with other studies, the individual-specific perceptions and attitudes varied across the sample respondents, reflecting underlying heterogeneity in interpreting and understanding the risks posed by climate change and SLR (Hunter et al., 2012). This heterogeneity was most pronounced in RP across gender and income, and in RC across marital status, and income. Interestingly, there was no difference across sample respondents as far as RR attitude was concerned.

The models developed in this study demonstrate that the determinants of WTP for recreational ecosystem services in light of the risks posed by climate change and SLR are complex. While individual socio-demographic and economic factors tend to be important determinants of public preferences and WTP (Halkos & Matsiori, 2012), our study points to education as the single most influential socio-demographic variable (model 3, Table 3.4). The relationship between individuals' education and their responses to risk items were both positive and significant.

3.5.3 Implications for water resource management

Management and policy decisions involving complex ecological systems such as the Everglades are well served by employing an integrated framework combining natural and social sciences to achieve sustainable and welfare-optimizing solutions (Turner et al., 2000). This study provides water managers with insight into the associated economic benefits of improved fishery ecosystem services vis-à-vis increased freshwater inflows, particularly under the looming threat of SLR and its potential impacts on the Everglades ecosystems, and addresses some of the classic challenges of ecosystem management: uncertainty about the future need for action, lack of current political support, and financial burden. Results of this study indicate that the public believes the predicted SLR and attendant negative impacts on the Everglades will occur during their lifetime. Additionally, that they cannot live with the negative impacts suggests a desire for public agencies to act to mitigate the impacts, although results also suggest that people believe that agencies alone cannot solve the problem without some assistance from the public.

Results also indicate that respondents have a moderately high level of concern about the loss of freshwater and the ensuing effects on recreational fishery productivity and their recreational experience. This high level of concern regarding freshwater and SLR may contribute to why respondents overwhelmingly express a desire for some government action to be taken against these future risks, and why they also feel that they as resource users should be responsible for contributing toward future restoration efforts.

As mentioned previously, this study found the presence of strong risk perception and concern about climate change and SLR. Results also provide a clear indication of the significant influence of risk perception on users' valuation of ecosystem services. The authors used the hydro-bio-economic model developed by Brown et al. (2018) to estimate the impact of risk perception of valuation. This model allows for monetizing the ecosystem service benefits of restoring freshwater flow from the past monthly average levels (usually low) to their respective environmental flow targets (high). These benefits were estimated using RP-adjusted WTP values as well as non-RP-adjusted values (see Table 3.7). When the WTP values were adjusted for Risk Perception (Table 3.6), the annual overall ecosystem service valuation (benefit) of users was 40.03% higher than the annual benefits estimated using non-RP-adjusted WTP. Similarly, if WTP were held at the highest level of RP, the annual total ecosystem benefits would be 136.69% higher than the estimate without RP-adjusted WTP. Thus, if freshwater managers are to implement any management policies (e.g., water allocation or user fees for restoration purposes), it's worthwhile to consider the appropriate WTP estimates, particularly ones which are adjusted for risk perception. When ecosystem services are lost, individuals who are more risk concerned lose more in terms of the value they place on those key ecosystem services than they would otherwise (i.e., with very low or no perception or concern toward the risk).

The immediacy of the resource users' experience with the ecosystem services considered in this study (at least once in the past three years) lends itself to their having a higher degree of RP, RC, and RA, and consequently 40% higher WTP. Thus, these members of the public, the users of impacted ecosystem services, do not demonstrate a high degree of environmental neoskepticism, which could impact environmental managers and public agencies. In keeping with the classic "user pay principle" (Muradian, O'Connor, & Martínez Alier, 2002), managers and agencies should first go after the users who are more engaged and have a higher RP and in turn WTP, but should be cautious not to unfairly burden them with excessively high user fees. Ultimately, these ecosystem services are public goods, which need to be paid for by the general public, regardless of intensity of use.

3.6. Conclusion

Results of this study indicate that resource users attach positive and significant values to the Everglades recreational fisheries, and the higher levels of WTP suggest they would see an increase in the nonmarket benefits for improved restoration efforts. The impact of their risk perception, concern, and attitude toward risk reduction actions on WTP are particularly salient for water resource managers as they develop plans for future restoration in terms of public support for and public funding of these efforts. Public perception of freshwater decline and SLR, concern for future risk, and strong feelings of personal responsibility could all be significant drivers of stronger political support for restoration and mitigation actions by public agencies, including water allocation decisions that prioritize Everglades restoration, new user-based financing options, and programs to increase public awareness of how freshwater management addresses climate risks.

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Tables

Table 3.1. Respondent Demographics

	Sample (600)					
	n	Mean	Median	Std. Dev.	Min	Max
Age	597	38.31	35	14.53	18	85
Household size	594	2.90	3	1.31	1	10
Household income ($)	566	64,434	50,000	42,002	20,000	240,000
Married	n	%				
Yes	303	50.67				
No	295	49.33				
Gender	n	%				
Female	339	56.59				
Male	260	43.41				
Race/Ethnicity	n	%				
White	454	71.16				
Hispanic	91	14.26				
Black or African-American	55	8.62				
American Indian or Alaska Native	15	2.35				
Asian or Pacific Islander	10	1.57				
Other	13	2.04				
Education	n	%				
Less than high school	14	2.34				
High school / GED	118	19.70				
Some college	145	24.21				
College degree	226	37.73				
Graduate degree	63	10.52				
Professional degree	33	5.51				

Table 3.2a. Risk Perception, Concern and Attitude towards Risk Reduction

Perception of risks	***n***	**Mean**	**Std. Dev.**
(1) I believe that the predicted SLR will happen during my lifetime	599	3.57	1.10
(2) I can live with the negative impacts of SLR on the Everglades fisheries and my recreational experience	599	2.45	1.18
(3) It is too early to worry about SLR and the future generation will know how to handle the situation better	599	2.32	1.22
Concern about specific risks	***n***	**Mean**	**Std. Dev.**
(4) Regarding the SLR in the Everglades	600	3.90	1.07
(5) Regarding declining freshwater flow and increasing salinity	600	4.11	0.99
(6) For future fish abundance in the Everglades	600	4.23	0.93
(7) Regarding the overall ecosystem health	598	4.38	0.93
Risk Reduction or Regulation	***n***	**Mean**	**Std. Dev.**
(8) Government agencies must start to take actions to increase the freshwater flow in the Everglades	600	4.12	1.03
(9) As a citizen or resource user, I am also responsible for contributing towards the actions to minimize the impacts of SLR	600	4.07	0.96
(10) The relevant public agencies will manage the freshwater flow effectively without my contribution to the effort	597	2.77	1.25
(11) Any human activities that adversely influence the quality of the Everglades health should be regulated	600	4.05	0.99

Table 3.2b. Risk Perception, Concern and Reduction Preferences across Different Demographic Groups[1]

Demographic Characteristics	Levels	n	Risk Perception (Out of a Max Score of 15)	Risk Concern (Out of a Max Score of 20)	Risk Reduction (Out of a Max Score of 20)
Florida resident	Yes	570	10.8	16.6	15.5
	No	6	11.3	17.8	16.6
F			0.296	0.759	0.83
P			0.586	0.384	0.363
Married	Yes	295	11	16.7	15.4
	No	303	10.6	16.5	15.5
F			**3.892**	0.703	1.22
P			0.021	0.495	0.296
Gender	Female	260	10.6	16.3	15.4
	Male	379	11	16.9	15.5
F			2.056	**3.097**	1.475
P			0.129	0.046	0.23
Race	White	422	10.9	16.8	15.6
	Hispanic	85	10.5	16.4	15.2
	African American	55	10.6	16.2	15.1
	American Indian	14	11.3	15.9	14.8
	Asian or Pacific Islander	10	10.4	16.9	15.1
	Other	13	11	15.8	15.2
F			0.571	0.722	0.797
P			0.722	0.607	0.552
Education	Less than high school	12	10.2	15.4	15.5
	High school/GED	110	10.4	16.3	15.1
	Some college	136	10.6	16.5	15.5
	College degree	218	11	16.9	15.8
	Graduate degree	58	11.2	17.1	15.3
	Professional degree	30	11.3	16.9	15.3
F			**2.12**	1.052	0.848
P			0.062	0.386	0.516
Income[2]					
F			**2.145**	**1.72**	1.274
P			0.011	0.053	0.224

[1]In this table, the original respondents' scores of statements that reflect skepticism towards risk (statement # 2, 3 and 10 of Table 2a) 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 3.3. Results of Explanatory Factor Analysis

Shaded boxes show items loading on each factor with loadings > 0.5.

Variable	**Factor 1 (Positive Risk Perception)**	**Factor 2 (Risk Skepticism)**
Concern about SLR and its impact in the Everglades	0.7832	-0.0210
Concern about SLR and declining freshwater flow and increasing salinity	0.8409	-0.0428
Concern about SLR and its impact on future fish abundance in the Everglades	0.8237	-0.0967
Concern about SLR and its impact on overall ecosystem health	0.8073	-0.0453
I believe that the predicted SLR will happen during my lifetime	0.5206	-0.2866
Government agencies must start to take actions to increase the freshwater flow in the Everglades	0.7766	-0.1019
As a citizen or resource user, I am also responsible for contributing towards the actions to minimize the impacts of SLR	0.7266	-0.0738
Any human activities that adversely influence the quality of the Everglades health should be regulated	0.6674	-0.2352
I can live with the negative impacts of SLR on the Everglades fisheries and my recreational experience	0.3481	0.7180
It is too early to worry about SLR and the future generation will know how to handle the situation better	0.4009	0.6972
The relevant public agencies will manage the freshwater flow effectively without my contribution to the effort	0.1011	0.7731
Eigenvalue	**4.79053**	**1.76617**

Table 3.4. Results of Mixed Logit Model

	Model 1	Model 2	Model 3	Model 4	Model 5
ASC	2.5661*** (0.40324)	-5.0811*** (0.9900)	0.6338 (0.0268)	2.189*** (0.3955)	2.4309*** (0.3851)
Cost	-0.0200*** (0.0041)	-0.0181*** (0.0036)	-0.0203*** (0.0039)	-0.0185*** (0.0036)	-0.0204*** (0.0041)
Catch	0.0095*** (0.0037)	0.0085** (0.0032)	0.0095** (0.0036)		-0.0177 (0.0160)
Keeper	0.0015 (0.0049)	0.0024 (0.0044)	0.0025 (0.0049)		-0.511 (0.0226)
Distance	0.0006 (0.0030)	-0.00002 (0.0027)	0.0011 (0.0029)		0.0356** (0.0136)
EcoHealth	0.0174*** (0.0045)	0.0151*** (0.0038)	0.0172*** (0.0043)		-0.0472** (0.0174)
ASC*Perception		1.0753*** (0.2573)			
ASC*Concern		0.5912** (0.2450)			
ASC*Reduction		0.2960 (0.2292)			
ASC*Age			0.0268 (0.0179)		
ASC*Education			0.3847* (0.2315)		
ASC*Income			0.0199 (0.1563)		
Perception*Catch				0.0026** (0.0009)	0.0076* (0.0043)
Perception*Keeper				0.0014 (0.0012)	0.0145** (0.0061)
Perception*Distance				-0.0006 (0.0007)	-0.0097*** (0.0037)
Perception*EcoHealth				0.0052*** (0.0010)	0.0179*** (0.0049)
Observations	578	578	578	578	578
LR chi2	82.73	52.98	86.40	61.94	65.30
Log-likelihood	-1094.227	-1062.1834	-1082.9052	-1088.7186	-1066.0773

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

Table 3.5. Regression Variables

Variable	Description
ASC	Alternative specific constant
Cost	Additional per-trip cost
Catch	Catch rate
Keeper	Size of the largest keeper
Distance	Travel distance for fishing
EcoHealth	Overall ecosystem health
Perception	Perception of the risks posed by SLR and climate change
Concern	Level of concern about specific risks
Reduction	Attitudes and preferences toward risk reduction and regulation
Age	Respondent age (in years)
Education	Level of education
Income	Annual household income

Table 3.6. Marginal WTP Results at Various Levels of Risk Perception

	Model 1 Attributes only Model ($)	Model 5 Attributes-Interacted with Risk Perception Model						
Risk Perception		$\overline{RP}$ = 3.6	1[a]	2[a]	3	4	5	Risk-weighted Average WTP[b]
Catch WTP	$0.47	$0.47	*$0.00*	*$0.00*	$0.25	$0.62	$1.00	$0.62
EcoHealth WTP	$0.87	$0.85	*$0.00*	*$0.00*	$0.32	$1.20	$2.07	$1.21

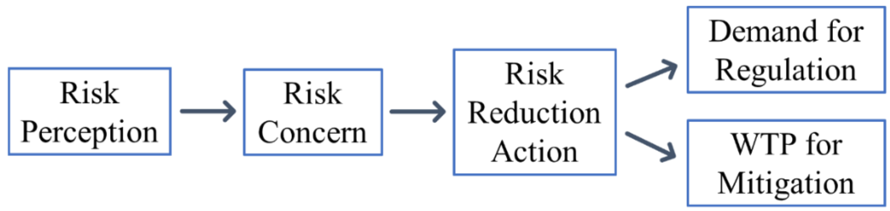
[a]Computed WTP values were negative at risk perception levels of Likert scale 1 and 2. Since negative WTP values (disutility from improved attributes) seem unrealistic, authors discard those values and assume WTP values to be zero at risk perception levels of 1 and 2.
[b]Risk-weighted average WTP values 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, 4.29% strongly disagreed (Likert scale = 1), 7.51% somewhat disagreed (2), 19.27% neutral (3), 31.88% agreed (4), and 37.04% strongly agreed (5) to the eleven risk questions.

Table 3.7. Effect of accounting for risk perception on the anglers' estimated benefit from improved recreational services in ENP

	Estimated Recreational Benefits from Restoring Freshwater Flow at Quarterly Target Levels Using		
	Risk-unadjusted WTP	Risk-adjusted WTP	Highest Risk Perception Level WTP
Fish WTP ($/% additional catch)	0.47	0.62	1.00
EcoHealth WTP ($/% improvement)	0.85	1.21	2.07
Quarterly benefits:			
Dec - Feb	1,502,326	2,089,681	3,514,415
Mar - May	3,803,337	5,372,410	9,139,190
Jun - Aug	704,106	975,280	1,635,025
Sep - Nov	845,053	1,161,589	1,936,027
Sum	6,854,822	9,598,960	16,224,657
Difference between risk perception-adjusted and unadjusted ($)		2,744,138 (40.03%)	9,369,835 (136.69%)

Figures

Figure 3.1. Psychometric risk perception-concern-mitigation paradigm adapted from Fischoff et al. (1978)



4. A Game-Theoretic Model of Crop Flood Indemnity in South Florida

4.1. Introduction

Changes in the frequency, spatial distribution, and magnitude of several climatic conditions and extreme events are likely to occur in the not too distant future and could pose significant risks to human well-being (IPCC 2014). Among such changes are an increased potential of flooding due to increased heavy precipitation events and accelerated sea level rise, posing particular concern to coastal communities and agricultural production. South Florida is among the areas of the U.S. most vulnerable to inundation (Gornall et al. 2010; Erwin 2009; Dolan and Walker 2006; Scavia et al. 2002). In addition to inundation, rising sea level can increase salinity of freshwater ecosystems and aquifers (Scavia et al. 2002). A mosaic of urban settlements, agricultural areas, and natural areas, South Florida is served by a highly human-engineered water management system (Harwell et al. 1996). Management agencies grapple with managing water to meet multiple objectives, including urban and agricultural water supply, flood control, and environmental restoration. Climate-induced (e.g., flood, drought, sea level rise) water shortage or excess often tests the limits of this engineering system, and extreme events are in turn expected to further increase the complexity of managing water resources for competing users.

Heavy precipitation and flooding events in the United States and worldwide in recent years have greatly damaged crop production. If model projections of increased weather extremes are realized (National Park Service 2009; IPCC 2014), the cost of crop losses could increase drastically. Recent studies have attempted to simulate the amount of plant damage from excess soil moisture in order to estimate crop production loss, and found that these losses under current climatic conditions might double in the next thirty years to an estimated $3 billion annually (Rosenzweig et al. 2002). In 2017, up to 7,000 acres of agricultural land in the southern portions of Florida experienced storm surge with salt water inundation during Hurricane Irma, with the Florida Department of Agriculture and Consumer Services estimating losses at over $30 million (Alvarez 2017). The costs of this and other losses may be borne directly by the farmers impacted or transferred to private insurers or governmental disaster relief programs.

As the expected level and intensity of flood and heavy precipitation events increase, the amount of indemnities paid upon losses due to these events would also increase. Thus, crop insurance claims can serve as a metric of the climate-related vulnerability of agriculture. To develop such a metric, it is necessary to study how crop insurance decisions are made. Participation patterns have shifted as new insurance products have expanded farmers' choices of types and levels of coverage, and the literature shows a variety of factors influencing farmers' choices among available crop insurance products (Makki and Somwaru 2006, 2001; Sherrick et al. 2004; Smith and Baquet 1996; Moschini and Hennessy 2001). Chief among these factors is the level of risk, followed by the cost of insurance, and the level of premium subsidy.

Frameworks for evaluating farmers' crop insurance decisions typically employ the standard assumption that farmers will maximize the utility of their net revenue subject to physical and technical constraints (Bar-Shira, Just, and Zilberman 1997; Sherrick et al. 2004; Smith and Baquet 1996; Shaik and Atwood 2017; O. Mahul 1999; Keith H. Coble and Knight 2002). These studies show that the levels of insurance premium and government subsidy are the two key determinants of farmers' participation. Nevertheless, these two rates are policy decisions made by Risk Management Services (RMS) of USDA each year. RMS grapples with the actuarial decision of optimizing insurance and subsidy rates such that net insurance premium creates enough incentive for farmers to purchase insurance protection, while private crop insurers are able to indemnify crop losses year after year adequately. This would require that either premiums, farmers' participation, or both are high enough to generate enough premium income to cover losses. However, farmers' participation level varies inversely with the premium. Furthermore, as the expected level and intensity of flood and heavy precipitation events increase, the amount of indemnities paid upon losses due to these events would also increase. Therefore, ultimate solvency of crop insurance market and climate-related crop risk reduction depend on the interactive decisions of RMS, farmers, and private insurers, under increasing level of climate-induced crop perils.

This paper links the occurrence of flooding events and crop insurance indemnity claims by simulating farmers' decision behavior of whether to purchase crop insurance and their choices among alternative products, considering varying risk of perils due to climate change and sea level rise. Government agricultural policy development and farmer response are modeled as a hierarchical Stackelberg leader-follower game-theoretic decision process (Bhat, Alexander, and English 1998; Bulut 2017). Hierarchical games are multi-level games with at least two players at each level. Such games can be either cooperative, in which negotiation between players is permitted, or non-cooperative, in which players make decisions independently. In our model, government is assumed to be the dominant player, or leader, choosing an optimal crop insurance premium and subsidy in order to optimize the participation response of the subordinate player, or follower, in this game represented by farmers. Ultimately, this paper seeks to understand the implications of funding insurance subsidies, how subsidy policies can influence participation rates, and setting premium prices for adequate participation by farmers and on private insurers' underwriting ability. The model is applied to two specialty crops in South Florida, which is predominant in sub-tropical agriculture, prone to tropical storms, and one of the least studied regions from a crop insurance point of view.

4.1.1. Crop Insurance

Due to a host of stochastic factors, including climatic conditions, agricultural production and specifically crop production has been characterized as volatile and risky (van Asseldonk, Meuwissen, and Huirne 2003; Joy Harwood et al. 1999). Dismukes (2002) states that "its economic returns are subject to events beyond a farmer's control," citing the examples of market conditions as well as rainfall, temperature, other weather conditions, plant disease, and insect infestations. The U.S. government has instituted various policies and programs in an attempt to assist farmers in managing these risks, starting as early as the 1930s, making payments to farmers in times of low prices and providing disaster assistance payments and crop insurance (Glauber and Collins 2002; Smith and Goodwin 2006; Dismukes 2002).

Since the 1930s, the federal crop insurance program has been an important agricultural policy instrument (Serra, Goodwin, and Featherstone 2003). The U.S. Federal Crop Insurance Corporation (FCIC) operates the federally subsidized Multiple Peril Crop Insurance (MPCI) Program. The 1980 Federal Crop Insurance Act overhauled the MPCI program with the intention to reduce the need for large-scale ad hoc disaster relief programs, expanding coverage access across crops and regions and establishing a target participation rate of 50% of planted acreage (Smith and Baquet 1996). The Federal Crop Insurance Reform Act of 1994 and the Agricultural Risk Protection Act of 2000 again restructured the crop insurance program, increasing premium subsidies mainly at lower insurance coverage levels then at higher coverage levels, respectively. The Food, Conservation, and Energy Act of 2008 further increased premium subsidy rated for certain insurable units of land. According to Collins and Bulut (2011), these progressive increases in premium subsidies stimulated higher and more diverse participation over time, improving the MPCI program's actuarial performance by "reducing adverse selection and enhancing underwriting and ratemaking." According to Coble (2002), high participation rate reflects farmers' acceptance of MPCI, and "has been a priority of policy makers and program administrators… [due to] the widely held belief that MPCI cannot effectively substitute for other forms of federal crop disaster relief unless a large proportion of farmers are insured."

Bulut (2017) cites four reasons for government support of crop insurance. Because crop insurance risks are systemic, they may result in missing markets (Duncan and Myers 2000). This discourages government use of ad hoc disaster payments, which discourage the purchase of crop insurance (United States Government Accountability Office 2014; van Asseldonk, Meuwissen, and Huirne 2003; Innes 2003). Additionally, asymmetric information, causing moral hazard and adverse selection problems, can lead to underinsurance market failure (Nelson and Loehman 1987). Lastly, farmers may be optimistically biased, systematically underestimating the risk of losses such as those caused by natural disasters (D. R. Just 2002; Oliver Mahul and Stutley 2010; K. H. Coble and Barnett 2013).

Knight and Coble (1997) undertook a survey of crop insurance literature from 1980 through 1997, examining econometric research conducted at both farm and aggregate levels. Farm-level models of demand for crop insurance, following Calvin (1992), typically strive to explain farmers' decision of whether to purchase insurance, or, following Smith and Baquet (1996), study the decisions of both whether to purchase and how much to insure. When taking an aggregate approach, such as at the county- or state-level, as in Goodwin (1993), studies generally seek to explain the proportion of either land coverage or farmers choosing to insure. Recent studies, such as Richards (2000), have examined not only how much land a farmer chooses to insure but also the level of coverage they purchase.

Studies at the farm level have taken a variety of approaches to incorporate some measure of return to or cost of insurance. For example, while Calvin (1996) and Coble et al. (1996) incorporated the expected return to insurance, Just and Calvin (1990) calculated the quasi-rent resulting from insurance. Both Goodwin and Kastens (1993) and Smith and Baquet (1996) follow Goodwin (1993) in incorporating the MPCI premium, as does this study. These previous studies using premium rate found negative effects on participation and quantity of coverage.

While our study incorporates the MPCI premium to analyze participation, we do so using aggregate county-level data to estimate MPCI participation, following similarly to Gardner and Kramer (1986), and Barnett, Skees, and Hourigan (1990). Gardner and Kramer (1986) found that at the aggregate level, the expected rate of return to insurance [(expected indemnity – premium) / (premium)] had a significant positive effect on participation. This is consistent with Cannon and Barnett, who found a negative effect of change in the net cost of insurance on crop insurance participation.

Studies about MPCI purchase decisions since Knight and Coble's 1997 survey have tended to focus on a host of other factors, including risk characteristics and farm income level (Makki and Somwaru, 1999), revealing a relationship between risk and choice of insurance contract. Makki and Somwaru (2001) went on to analyze longitudinal crop insurance purchase decision data from 1995 to 1999, identifying factors that influenced farmers' choices of crop insurance purchase to varying degrees: risk level, premium price, premium subsidy, expected indemnity payoffs, availability of alternative insurance products, and various insurance contract characteristics. A study by Changnon (2004) on the effects of drought forecasts on crop insurance decisions, based on a survey of Midwestern (Illinois, Indiana, Iowa, Nebraska, and Ohio) farmers, found that 40% of respondents changed their coverage based on the drought forecasts. In terms of risk characteristics, Sherrick et al. (2004) found that farmer-specific reservation insurance premiums depended on expected rates of return with and without insurance, as well as farmer-specific levels of risk aversion. Considering data from a 1989 survey, Just, Calvin, and Quiggin (1999) concluded that "risk aversion is a relatively weak incentive for participation… [suggesting] that farmers' asymmetric informational advantages lead to insuring those operations with higher expected indemnities" (Ramirez and Scott Shonkwiler 2017).

An analysis of the effect of increased insurance subsidies on land use by Claassen, Lubowski, and Roberts (2005) determined that in some areas, as crop insurance subsidies rose, a greater amount of land was put into production. A study by Babcock and Hart (2005) specifically examined what effect the subsidy changes related to the 2000 Agriculture and Risk Protection Act (ARPA) had on insurance purchase decisions, comparing coverage levels before and after ARPA and concluding that purchases of coverage levels above 65% more than doubled. More recent studies examine the effect and magnitude of premium subsidies on insurance participation decisions. O'Donoghue (2014) found that corn insurance demand (measured as liability per acre) increases by 0.13%, nearly doubling depending on region, per percent increase in subsidy per acre. Similarly, Yu et al. (2018) found that a 10% increase in subsidy per dollar of liability induced a 0.43% increase in planted acreage.

Much of the literature on crop insurance focuses on participation. While this type of analysis provides insight about farmers crop insurance purchase decisions and behavior considering crop insurance subsidies, it fails to reveal much about the strategic interaction between government (who provides said subsidies) and farmers. More recently, however, efforts to analyze this in a theoretical framework have been undertaken (Bulut 2017). This study continues such efforts by examining the interaction with an empirical analysis of the cases of two specialty crops in South Florida, fresh market sweet corn and fresh market tomatoes. This study extends the crop insurance literature in several ways. It casts the three-way strategic interactions between farmers, insurers and government using a refined theoretical framework, Stackelberg leader-follower game. To our knowledge, this is the first empirical application of such model to crop insurance literature. Second, this study addresses a key policy relevant question that agricultural risk management agencies grapple with. Confronted with uncertain climate, farmers do need an affordable crop insurance program to keep their farming solvent. Unless there is adequate participation by farmers, insurance program can neither be affordable nor solvent. Their participation rate, in turn, directly depends on net premium they pay relative to their reservation premium (i.e., the maximum net premium they are willing to pay such that they are indifferent to purchase insurance or not). This reservation premium could depend on farmers' expectations about climate and business risks in general. Therefore, it is important for the risk management agency to set premium and subsidies such that they do not exceed farmers' reservation premium, in order to ensure a sustained farmers' participation in the program. The reservation premium and subsidy serve as respective ceiling and floor amount, respectively. The model presented in this develops a rule for empirically estimating such reservation values under varying climate risk scenario. Third, past research on insurance for specialty crops in the tropics is limited. South Florida specialty crops provide a unique geographic and insurance context given that the region is prone to increasing climate risk and crops involved are high value crops.

4.2. Methods

4.2.1. Delineation of the study area

This study focuses on two specialty crops, fresh market corn and fresh market tomatoes, in Miami-Dade and Palm Beach counties in South Florida. According to Ligon (2011), "Specialty crops, particularly fruits and vegetables, differ in several important respects from traditional commodity crops in ways which may affect both demand for insurance and the difficulty of supplying insurance." For example, spatial shocks (e.g., heavy precipitation events) which affect production within a relatively small geographical area will have a greater effect on aggregate supply than the same shock for a cereal crop, which is storable and has more geographically dispersed production (Ligon 2011).

The insured liability of specialty crops has been trending upward in recent years (Collins 2012). In 2011, specialty crops accounted for approximately 2.6 percent of total insured acres, but approximately 10.3 percent of total insured liability. This is due in part to specialty crops' high value per acre. Collins (2012) goes on to describe various challenges in expanding insurance coverage for specialty crops, including the small acreages of some specialty crops, accurately assessing the effects of weather on crop production for loss adjustment, and the usual insurance problems of adverse selection and moral hazard.

As of 2016, Florida ranked first in value of production of fresh market tomatoes, accounting for 40% of the total U.S. value, and second in value of production of fresh market sweet corn, accounting for 24% of the total U.S. value (Florida Department of Agriculture and Consumer Services 2017).

Having a subtropical to tropical climate with a wet (warm) season and a dry (cool) season, cropping season in South Florida for these vegetables typically coincides with the dry season of October through May. Precipitation patterns in general as well as extreme precipitation events (in South Florida are found to be significantly correlated with large-scale climate effects, including the Atlantic Multidecadal Oscillation (AMO), with a 55-70 year periodicity; the Pacific Decadal Oscillation (PDO), with a 20-30 year periodicity; and the El Niño Southern Oscillation (ENSO), with a 3-7 year periodicity (Gunn 2010; South Florida Water Management District 2011; Wong et al. 2014). During El Niño years, the polar jet stream takes a more southerly flow which allows more frontal systems to reach Florida, increasing precipitation particularly in the dry season (South Florida Water Management District 2011). For the study period of 1989 to 2017, El Niño years are 1991-92, 1994-95, 1997-98, 2002-07, 2009-10, and 2014-16 (NOAA). The 2014-16 El Niño was particularly strong compared to the rest of the US states (Figure 1), leading to nearly $3 million in sweet corn and tomato losses in Miami-Dade and Palm Beach counties.

4.2.2. Game-theoretic models

Game-theoretic methods, both cooperative and non-cooperative, have been widely used to simulate the strategic behaviors of agents in the contexts of manufacturing (Zhao et al. 2012), engineering (Liu, Ji, and Jiao 2013), environmental policy (Bhat, Alexander, and English 1998; Sinha et al. 2013; Hong et al. 2017), and

in insurance markets, e.g., natural disaster and crop insurance markets (El-Adaway et al. 2015; O. Mahul 1999; Bulut 2017). Hierarchical market solutions, first introduced by Heinrich von Stackelberg in 1934, have been employed to simulate sequential decision-making in situations in which one agent has dominating power over the others (von Stackelberg 1952). Now known as a Stackelberg equilibrium, this sequential game solution concept involves players with asymmetric roles, one a leader and the other following. The leader announces their action and the follower responds by choosing their optimal response given that announcement. The leader, knowing the follower's objective function and anticipating the response, chooses the action that optimizes their own performance given the follower's rational response. Followers, in order to maximize their own objective function, decide whether to take certain action. In a crop insurance market, for instance, farmers as followers would decide which insurance product to purchase given the price of insurance premium, their risk factors, and a vector of economic variables. Government, knowing farmers' optimal decision, sets the premium and subsidy levels. The farmers' dynamic problem will first be developed, and all necessary conditions derived. These conditions will then be included as constraints in the development of the government's dynamic problem, in which it attempts to balance the income from and flood indemnity claims paid to farmers' insured crops by insurers. Interaction between government and farmers is assumed to be non-cooperative, which can still result in a socially efficient decision strategy under certain possible conditions (Bhat, Alexander, and English 1998). To this end, a sequential hierarchical game becomes particularly relevant for simulating the decision behavior of the government as a price-setter for both premiums and subsidies, and farmers as followers aiming to maximize their profits given the likelihood of perils.

4.2.2.1 The farmers' model

We consider a large number (N) of farmers. The farmers' objective is to maximize net revenue from agricultural production (market return less production cost), subject to stochastic peril. Following Duncan and Myers (2000), a typical farmer *n* of the population faces the prospect of a loss with probability θ and no loss with probability $(1-\theta)$. Farmers are assumed to lack the ability to influence the government's policy decision once it is made. Alternatively, they attempt to optimally make their decisions regarding insurance purchase in response to the government's decision variable. It is assumed that farmers are price takers and are in a climatologically homogenous region. Without the purchase of insurance, an individual farmer *n*'s total net revenue (*TR*) is

$$TR = A(1-\theta)R \tag{1}$$

where A is planted acres, R is net revenue per acre, and θ is the expected probability that a certain peril will occur. With the purchase of insurance, the farmer's *TR* will be lowered by the net premium paid as below,

$$TR_j = A[\theta R_j - (P_j - S_j)] \tag{2}$$

where P is the cost of insurance premiums, S is the subsidy for purchase of insurance, and subindex j is the specific insurance product purchased by the farmer. Following the crop insurance decision framework of Sherrick et al. (2004), a farmer will decide to purchase insurance product j if their expected *TR* (or utility) with insurance is at least as much as without the insurance. Formally,

$$Prob(yes\ to\ j) = Prob\{A[\theta R_j - (P_j - S_j)] \geq A(1-\theta)R\} \tag{3}$$

The above inequality can be solved for either the farmer's maximum willingness to pay (WTP) for insurance (P_j^*) or the minimum subsidy (S_j^*) a farmer is willing to accept (WTA) to participate in the insurance market. Solving the inequality, the probability that $(TR - TR_j) \geq 0$ is assumed to yield a linear probability function,

$$Prob(\Delta TR_j \geq 0) = \alpha X \tag{4}$$

where X is a vector of P_j, S_j, M_j, and *CCI*, and where M represents prior participation and *CCI* represents general market conditions. Logit and probit models were also considered, and ultimately rejected in favor of a linear probability model using a censored Tobit estimator.

4.2.2.2 The government's model

Adapting Bulut (2017), a two-stage strategic interaction between the government and farmers is considered. In the first stage, the government announces the *ex ante* premium and subsidy rates. In the second stage, farmer *n* makes their insurance purchase decision by taking these rates as given. Following this, stochastic events unfold, and a loss does or does not occur. Thus, in this scenario, the government is the natural Stackelberg leader, making the farmer the follower. In the first stage, the leader (government) solves the problem of setting premium and subsidy rates by determining how the follower (farmers) will respond in the second stage.

With symmetric information, the government, the leader of this game, is assumed to set the price of the premium at the optimal participation price P_j^*, knowing that the farmer-followers will optimally decide their participation rate as in (Eq. 4) in response to the leader's optimal insurance rate decision and given level of subsidy. That is, the leader attempts to set the premium P_j^{**} at a rate that implicitly equates the total premium payment with expected indemnity payment. Formally, P_j^{**} can be determined by solving,

$$C_j P_j - C_j \theta R_j = C_j\left(P_j - \theta R_j\right) = 0 \tag{5}$$

Subject to (Eq. 4), where C_j is the extent of area covered by insurance and is a function of $Prob(yes\ to\ j)$, or $Prob_j^*$ and total acres ($\bar{A}$).

That is, $\bar{A}Prob_j^*(yes\ to\ j)\left(P_j - \theta R_j\right) = 0$, or,

$$\bar{A}Prob_j^*\left(P_j, S_j, M_j, CCI\right)\left(P_j - \theta R_j\right) = 0 \tag{6}$$

From (6) above, we can develop a function for the optimal premium P_j^{**} which will then determine the optimal participation rates of farmers for insurance product j. P_j^{**} is the most that farmers would be willing to pay (reservation premium) at which their return with and without insurance would be the same. Formally,

$$P_j^{**} = \beta_0 + \beta_1 I_{t-1} + \beta_2 CropPrice + \beta_3 RainEvent \tag{7}$$

4.2.3. Estimation of willingness to pay and willingness to accept

Probability of participation (Eq. 4) was estimated as a censored Tobit model, with a lower limit of 0 and upper limit of 1. Following standard practice (Johnston et al. 2013), the estimates of WTP ($P*$) and WTA ($S*$) were expressed as the ratios of the variable coefficients to participation coefficient as in (Eq. 4). Formally,

$$P^* = \frac{\hat{\beta}_0 + \hat{\beta}_2 \bar{S} + \hat{\beta}_3 \bar{M} + \hat{\beta}_4 \widehat{CCI}}{\hat{\beta}_1} \tag{8}$$

And,

$$S^* = \frac{\hat{\beta}_0 + \hat{\beta}_1 \bar{P} + \hat{\beta}_3 \bar{M} + \hat{\beta}_4 \widehat{CCI}}{\hat{\beta}_2} \tag{9}$$

Crop insurance and loss data were retrieved from the United States Department of Agriculture (USDA) Risk Management Agency. Crop production and value data were retrieved from the USDA Census of Agriculture. Sources of weather and climate data include the National Oceanic and Atmospheric Administration (NOAA) National Centers for Environmental Information Climate Data Online and NOAA National Weather Service Climate Prediction Center. Data were aggregated into a county-level format for each of the two selected crops, and monthly and annual means were calculated for the study period. Dollar variables are deflated to 2017 values. All analysis was conducted using Stata13.

4.3. Results and analysis

4.3.1 Participation

At sample mean levels, estimated likelihood of participation for farmers growing fresh market sweet corn was 0.7523. Estimates for likelihood participation were higher in Miami-Dade County than in Palm Beach County, at 0.7821 and 0.7278, respectively.

Farmers growing fresh market tomatoes were approximately half as likely to participate in the insurance market, with overall likelihood at 0.3485. Similarly to sweet corn farmers, Miami-Dade tomato farmers had a higher likelihood of participation than those in Palm Beach, at 0.4041 and 0.2498, respectively. Results from Tobit model estimations of insurance market participation are described in Table 4.1.

Coefficients for premium and subsidy were significant and with expected signs, and for both sweet corn and tomatoes the subsidy coefficient is nearly twice that of the premium, indicating that the level of government subsidy is a stronger driver of insurance purchase decisions than the premium price. Subsidies are on average 55% and 58% of the premium price over the study period for sweet corn and tomatoes, respectively, and the model reflects this relationship. Insurance participation in the previous period was positive and significant, and for both sweet corn and tomatoes was the most influential variable. The coefficient for CCI was also positive and significant, although more than double for tomatoes versus sweet corn.

The sample mean per-acre premium price across the study period of 1995 to 2017 for fresh market tomatoes was $444.56 per acre (in 2017 U.S. dollars), more than quadruple that of fresh market sweet corn ($100.21 per acre). Sample mean of subsidy in dollar per acre for sweet corn and tomatoes, deflated to 2017 U.S. dollars, are $55.49 (55.37% of the average premium) and $258.99 (58.26% of the average premium), respectively.

For fresh market sweet corn, farmers' maximum WTP for premium was $401.09 in Miami-Dade and $264.03 in Palm Beach, and $325.84 across both counties. Minimum subsidy WTA was $29.77, $86.15, and $60.72 across Miami-Dade, Palm Beach, and both counties, respectively. The percent of reservation subsidy to premium in Miami-Dade was 7.42% and in Palm Beach was 32.63%, and 18.63% overall across both counties. Farmers of fresh market tomatoes similarly had a higher WTP for premium in Miami-Dade than in Palm Beach, at $1023.35 versus $907.14, and $962.12 overall. Miami-Dade tomato farmers were WTA a minimum subsidy of $91.33, while in Palm Beach the minimum was $0. The percent of reservation subsidy to premium was 8.92% in Miami-Dade and 0% in Palm Beach, and 3.11% overall.

4.3.2 Leader's actuarial premium model estimation

Various model specifications for each fresh market sweet corn and fresh market tomatoes are described in Table 4.2. Variables in the models reflect those that RMA considers for its derivation of actuarially fair premiums, including expected crop price, expected participation, expected losses, and expected peril (in this case the specific peril is extreme precipitation). The optimal fresh market sweet corn premium estimated at the sample average using Model C2 was $126.79. Using Model T1, the estimated per-acre premium for maximum participation of fresh market tomato farmers was $508.32.

Maximum WTP and minimum WTA were estimated for premium and subsidy, respectively, for each county individually as well as for sweet corn and tomatoes overall and are described in Table 3.

At the reservation premium, or maximum WTP for insuring sweet corn, there is only one crop year, 2010, in which per-acre indemnity exceeds the per-acre reservation premium (Figure 4.2). However, the per-acre indemnity exceeds the average annual actual crop premium several years during the study period. On the contrary, the actual crop premium stays much below the maximum reservation premium. For tomatoes, the per-acre indemnity twice exceeds the maximum per-acre premium, in 2002 and 2012 (Figure 4.3). The percent of reservation subsidy with respect to premium for sweet corn and tomatoes overall were 18.63% and 3.11%, respectively. In Miami-Dade, the percent of subsidy was 7.42%, compared to Palm Beach's 32.63%. This suggests that Palm Beach farmers demand higher subsidy rates than do Miami-Dade farmers, and Miami-Dade farmers may be more risk-averse. For tomatoes, however, this is reversed, with Miami-Dade farmers demanding a higher subsidy rate of 8.92% to 0% for Palm Beach, indicating that Palm Beach farmers are willing to insure even without subsidies.

4.3.2 Policy and climate simulations

The U.S. Government Accountability Office (2014) found that reducing premium subsidies could potentially save hundreds of millions of dollars in the federal budget, Tables 4 and 5 illustrate the effects this action would have on participation rates. Holding per-acre premium prices constant, reducing subsidies by 20% lowers participation from 75% to 68% and from 35% to 28% for sweet corn and tomatoes, respectively, at the highest level of reduction (Table 4). Nearly all years in which per-acre indemnity exceeds per-acre premiums coincide with occurrences of El Niño (Figures 4.2 and 4.3).

4.4. Conclusions

The existing literature primarily looked at farmers' participation behavior (i.e., response to insurance premium or subsidy), without looking at the policymaker's decision. Our paper is the first to empirically capture the simultaneity of farmers' and government decisions through a hierarchical strategic leader-follower game model. Farmers' crop insurance participation decision was driven mainly by prior participation and the levels of premium and subsidy. The government's decision was affected by participation levels, crop price, prior indemnity, and peril, and varied by county. The models yielded optimal WTP for premium and WTA for subsidy, which have been above the government-set premium rates.

Actual indemnities from flood and excess moisture have exceeded crop premiums in several years, and maximum WTP in one year, while per-acre premiums consistently stayed well below maximum WTP.

As evidenced in Figures 4.2 and 4.3, mean per-acre indemnity resulting from excess moisture or precipitation sometimes exceeded the mean per-acre premium for both sweet corn and tomatoes, while the per-acre premium has remained consistently well below farmers' maximum WTP. The RMA has been able to set the premium at a low enough price to encourage participation, while also keeping subsidy rates at a fairly consistent level, which has caused the per-acre indemnity to exceed the actual premium a number of times. This demonstrates that the government must be cautious in setting premiums in response to expected perils. Any government decision to reduce subsidies may adversely affect the farmers' decisions and destabilize the overall crop insurance market. Particularly as climate risks continue to increase, this situation may not sustain the crop insurance market in the long term, and since the reservation premiums for both sweet corn and tomatoes are much higher than the actual observed premiums, farmers may be able to handle higher premium prices than what they are currently paying. However, that means that RMA must bear a higher subsidy burden, which may be an inevitable policy choice RMA may have to make in an effort to keep the insurance market afloat.

A trend analysis by the South Florida Water Management District (2011) from 1950-2008 shows a general decrease in wet season precipitation, possibly due to a shortening or delay of the wet season, but an increase in the number of wet days during the dry season. Coupled with sea level rise and the attendant need for flood control, a series of strong El Niño years could prove catastrophic not only for farmers but for insurers. Reductions in subsidies may also negatively impact insurers. As subsidy rates decline, so do the corresponding participation rates for a given insurance product.

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Tables

Table 4.1. Tobit regression results, insurance market participation in Miami-Dade and Palm Beach counties

	Fresh Market Sweet Corn	**Fresh Market Tomatoes**
Observations	51	45
Log-Likelihood	44.0313	31.5504
Variable	**Coefficient (P-value)**	**Coefficient (P-value)**
Per-Acre Premium	-0.0033 (0.00)	-0.0006 (0.03)
Per-Acre Subsidy	0.0064 (0.00)	0.0012 (0.05)
Previous Year Participation	0.5499 (0.00)	0.7343 (0.00)
Consumer Confidence Index	0.0017 (0.02)	0.0037 (0.00)
Constant	0.1151 (0.30)	-0.3473 (0.02)

Table 4.2. Premium regression results

Fresh Market Sweet Corn				
	Model C1	**Model C2**	**Model C3**	**Model C4**
Adjusted R Square	0.93	0.87	0.88	0.22
Observations	36	36	36	36
Variable	**Coefficient (Std. Error)**	**Coefficient (Std. Error)**	**Coefficient (Std. Error)**	**Coefficient (Std. Error)**
Intercept		-42.5014 (65.6348)	-36.7346 (61.8134)	-166.8853 (155.1799)
Per-Acre Indemnity$_{t-2}$	0.06236 (0.03778)	0.06248 (0.03814)	0.06207 (0.03755)	0.26203 (0.08745)**
Crop Price$_{t-1}$	0.0384 (0.0080)*	0. 0452 (0. 0132)*	0.04592 (0.0128)**	0.04896 (0.0327)
Extreme Rain Event$_{t-1}$	0.10545 (0.9153)	0.2892 (0.9666)		
Participation$_{t-1}$	53.18154 (21.8593)**	72.8229 (37.5099)**	70.4319 (36.1067)**	126.1255 (91.1857)
County Dummy (PB=1)	-123.7143 (9.683)*	- 123.3320 (9.7931)*	-122.4839 (9.2354)*	
Fresh Market Tomatoes				
	Model T1	**Model T2**	**Model T3**	**Model T4**
Adjusted R Square	0.73	0.74	0.01	0.04
Observations	36	36	36	36
Variable	**Coefficient (Std. Error)**	**Coefficient (Std. Error)**	**Coefficient (Std. Error)**	**Coefficient (Std. Error)**
Intercept			790.0440 (311.0084)**	811.7306 (291.4989)
Per-Acre Indemnity$_{t-2}$	0.1537 (0.0902)***	0.16188 (0.0836)***	0.1138 (0.0796)	0.1081 (0.0743)
Crop Price$_{t-1}$	0.02007 (0.0167)	0.02398 (0.0081)*	-0.0264 (0.02123)	-0.0266 (0.0209)
Extreme Rain Event$_{t-1}$	2.4297 (9.0430)			
Participation$_{t-1}$	104.85 (174.2490)	93.67687 (166.7427)	-57.4941 (165.2168)	-76.3857 (140.5113)
County Dummy (PB=1)	80.6564 (97.4179)	89.0560 (90.9182)	20.0152 (88.3249)	

**p<.01, **p<.05, ***p<.10*

Table 4.3. Maximum premium WTP and minimum subsidy WTA for participation in crop insurance market, 2017 U.S. $

	Overall	**Miami-Dade**	**Palm Beach**
Fresh Market Sweet Corn			
Maximum premium WTP	$325.84	$401.09	$264.03
Minimum subsidy WTA	$60.72	$29.77	$86.15
Percent of reservation subsidy to premium	18.63%	7.42%	32.63%
Fresh Market Tomatoes			
Maximum premium WTP	$969.12	$1023.35	$907.14
Minimum subsidy WTA	$30.06	$91.33	$0
Percent of reservation subsidy to premium	3.11%	8.92%	0%

Table 4.4. Participation at various levels of subsidy reduction for Sweet Corn in Miami-Dade and Palm Beach Counties, 2017

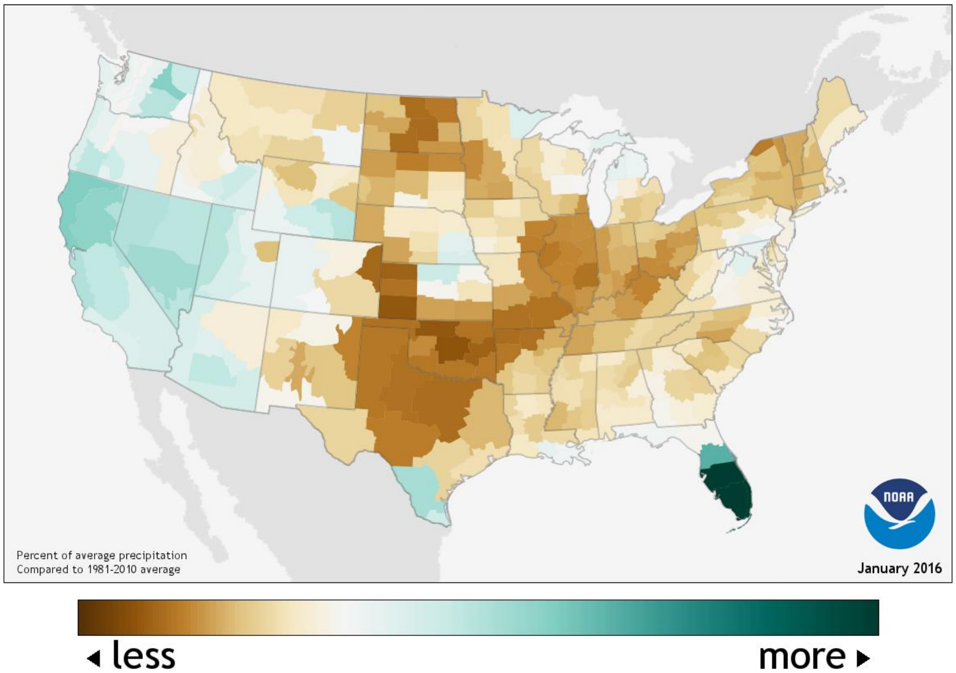
Fresh Market Sweet Corn				
% Reduction in subsidy	**5%**	**10%**	**15%**	**20%**
$/acre Reduction in subsidy	*2.77*	*5.54*	*8.32*	*11.09*
New subsidy per acre	$52.71	$49.94	$47.16	$44.39
Participation rate	73.43%	71.64%	69.84%	68.04%

Table 4.5. Participation at various levels of subsidy reduction for Tomatoes in Miami-Dade and Palm Beach Counties, 2017

Fresh Market Tomatoes				
% Reduction in subsidy	**5%**	**10%**	**15%**	**20%**
$/acre Reduction in subsidy	*12.94*	*25.89*	*38.84*	*51.79*
New subsidy per acre	$246.04	$233.09	$220.14	$207.19
Participation rate	33.29%	31.72%	30.16%	28.61%

Figures

Figure 4.1. Percent of average precipitation compared to 1981-2010 average, January 1996



(Source: https://www.climate.gov/maps-data/data-snapshots)

6Figure 4.2. Per-acre indemnity vs. premium, Fresh Market Sweet Corn 1990-2017

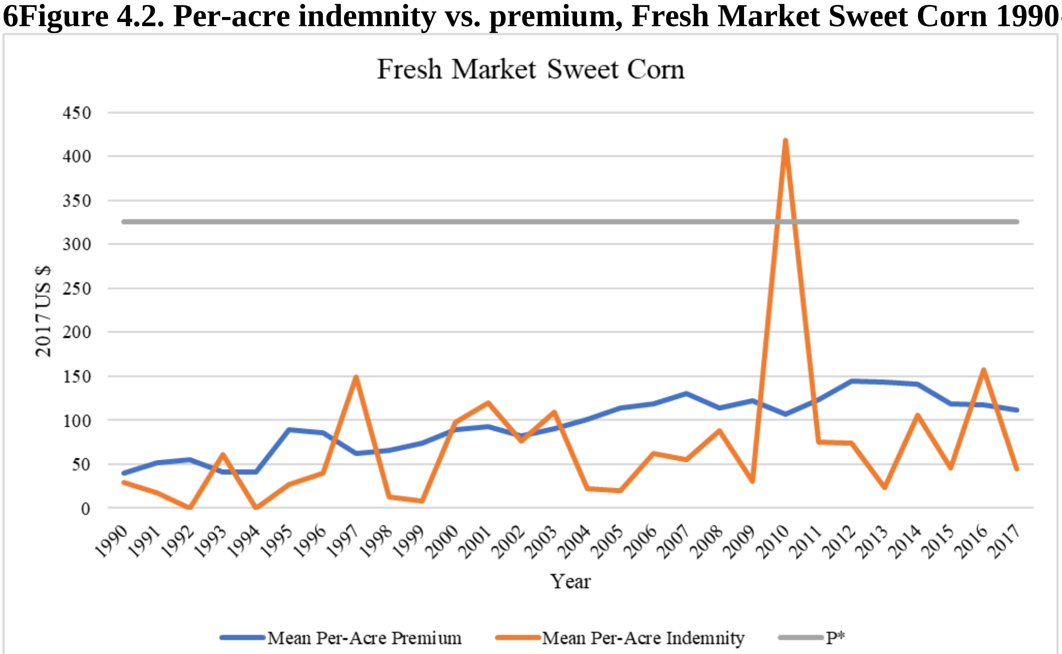
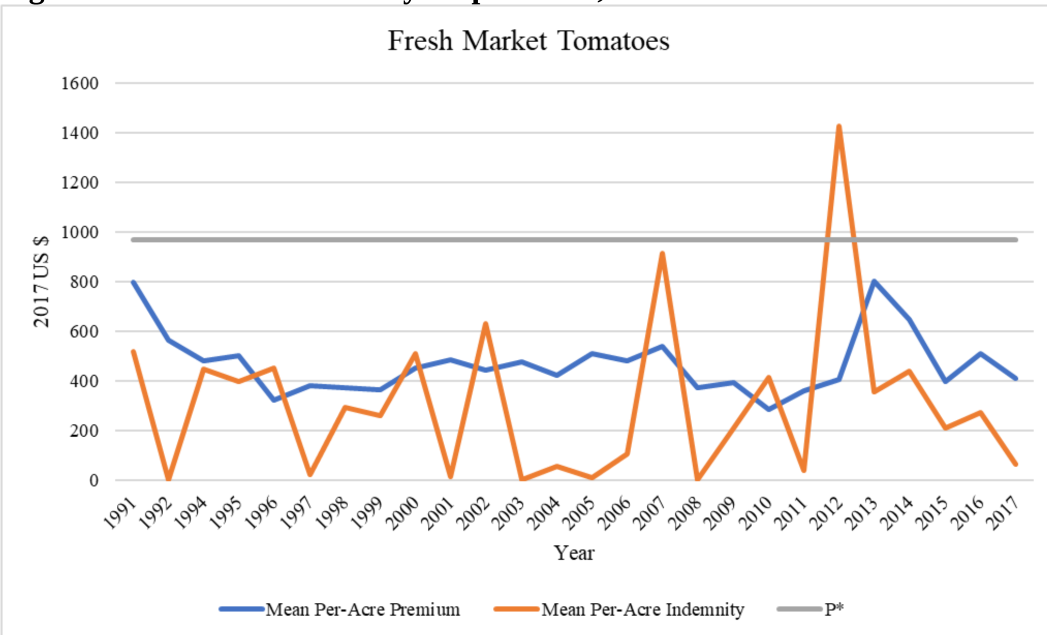


Figure 4.3. Per-acre indemnity vs. premium, Fresh Market Tomatoes 1991-2017



VITA

CHRISTINA ESTELA BROWN

	Born, Miami, Florida
2012	B.A., Economics Florida International University Miami, Florida
2018	M.S., Environmental Studies Florida International University Miami, Florida
2017-2019	Doctoral Candidate Florida International University Miami, Florida

PUBLICATIONS AND PRESENTATIONS

Brown, C. E. (April 2015). *Valuation of Fishery Ecosystem Services of the Everglades Water Management*. Poster presented at the Greater Everglades Ecosystem Restoration, Coral Springs, Florida.

Brown, C. E. (February 2016). *Modeling a Hydro-Economic Penalty Function for Recreational Fisheries in Everglades National Park*. Presented at the Florida International University Department of Earth & Environment Graduate Research Symposium, Miami, Florida.

Brown, C. E., Bhat, M. G., Rehage, J. S., and Sukop, M. (March 2016). *Linking Hydro-Ecological Relationships to Economics for Recreational Fisheries in the Coastal Everglades*. Poster presented at the annual Water Sustainability and Climate Investigator Meeting, Arlington, Virginia.

Brown, C. E. and Bhat, M. G. (June 2016). *An Integrated Framework for Linking the Ecosystem Services Valuation with the Freshwater Flow in the Florida Everglades*. Presented at The International Society for Ecological Economics, Washington, D.C.

Brown, C. E. (January 2017). *Integrating Economic and Biophysical Models for Ecosystem Services Valuation in the Florida Everglades*. Poster presented at the National Council for Science and the Environment, Washington, D.C.

Brown, C. E. (February 2018). *Risk Perception and Willingness to Pay for Everglades Recreational Ecosystem Services*. Presented at the Florida International University Department of Earth & Environment Graduate Research Symposium, Miami, Florida.

Brown, C. E. (March 2018). *Crop Flood Indemnity Claims in South Florida*. Presented at the Florida International University Agroecology Symposium, Miami, Florida.

Brown, C. E., Bhat, M. G., Rehage, J. S., Mirchi, A., Boucek, R. E., Engel, V., Ault, J. S., Mozumder, P., Watkins, W., and Sukop, M. (2018). *Ecological-Economic Assessment of the Effects of Freshwater Flow in the Florida Everglades on Recreational Fisheries*. Science of the Total Environment, 627: 480–93.

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Brown, C. E. and Bhat, M. G. (September 2018). *Valuing Ecosystem Services Under Climate Risk: A Case of Recreation in the Florida Everglades*. Presented at the International Society for Ecological Economics, Puebla, Mexico.

Brown, C. E. and Bhat, M. G. (November 2018). *Valuing Freshwater Ecosystem Services: A Missing Peace in the Restoration and Climate Change Debate on the Florida Everglades*. Presented at the Southern Economic Association, Washington, D.C.

Brown, C. E. and Bhat, M. G. (April 2019). *Linking Recreational Ecosystem Service Benefits with Freshwater Management in the Everglades*. Presented at the Greater Everglades Ecosystem Restoration, Coral Springs, Florida.