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Ecological-economic assessment of the effects of freshwater flow in the Florida Everglades on recreational fisheries

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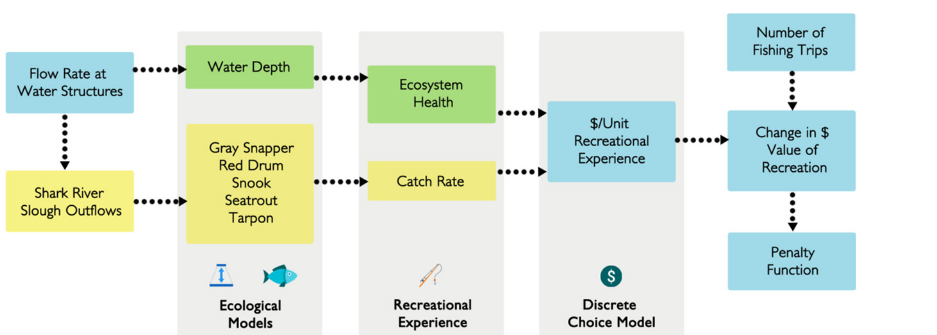
HIGHLIGHTS

- Develops an integrated methodology linking Everglades hydrology to economic values
- First ever estimate of anglers' willingness to pay for Everglades recreational experience
- Estimates losses in economic welfare due to missing freshwater delivery targets and implicit price of water use for recreation at \$41.54 AF^{-1}
- Relevant applications to management, restoration, and climate scenario analysis

GRAPHICAL ABSTRACT



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ABSTRACT

This research develops an integrated methodology to determine the economic value to anglers of recreational fishery ecosystem services in Everglades National Park that could result from different water management scenarios. The study first used bio-hydrological models to link managed freshwater inflows to indicators of fishery productivity and ecosystem health, then link those models to anglers' willingness-to-pay for various attributes of the recreational fishing experience and monthly fishing effort. This approach allowed us to estimate the foregone economic benefits of failing to meet monthly freshwater delivery targets. The study found that the managed freshwater delivery to the Park had declined substantially over the years and had fallen short of management targets. This shortage in the flow resulted in the decline of biological productivity of recreational fisheries in downstream coastal areas. This decline had in turn contributed to reductions in the overall economic value of recreational ecosystem services enjoyed by anglers. The study estimated the annual value of lost recreational services at \$68.81 million. The losses were greater in the months of dry season when the water shortage was higher and the number of anglers fishing also was higher than the levels in wet season. The study also developed conservative estimates of implicit price of water for recreation, which ranged from \$11.88 per AF in November to \$112.11 per AF in April. The annual average price was \$41.54 per AF. Linking anglers' recreational preference directly to a decision variable such as water delivery is a powerful and effective way to make management decision.

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This methodology has relevant applications to water resource management, serving as useful decision-support metrics, as well as for policy and restoration scenario analysis.

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. ENP 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 in., variation in tropical weather systems may result in wide seasonal variation and large year-to-year fluctuations (1901–2000 standard deviation of 11 in. in the Miami-Dade area) (Abtew 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., 2005a, b; 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 and Bachman, 2008).

As a large, subtropical estuary averaging in depth from 6 to 9 ft, Florida Bay provides critical habitat for a variety of species, including seagrasses and coastal mangrove communities (Bachman and 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. Recreational fishing in the Everglades 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 et al., 2005). These flows are regulated by the South Florida Water Management District (SFWMD) through massive canals and structures. The SFWMD determines monthly water delivery targets for the Everglades wetlands based on 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 >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 this 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 due to 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 this 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. Methods

2.1. Delineation of the study area

The geographic focus of the study is the ENP watershed, in particular the Shark River Slough (SRS) (Fig. 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 et al., 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 (Fig. 1). Lostmans River contributes 33% of mean annual

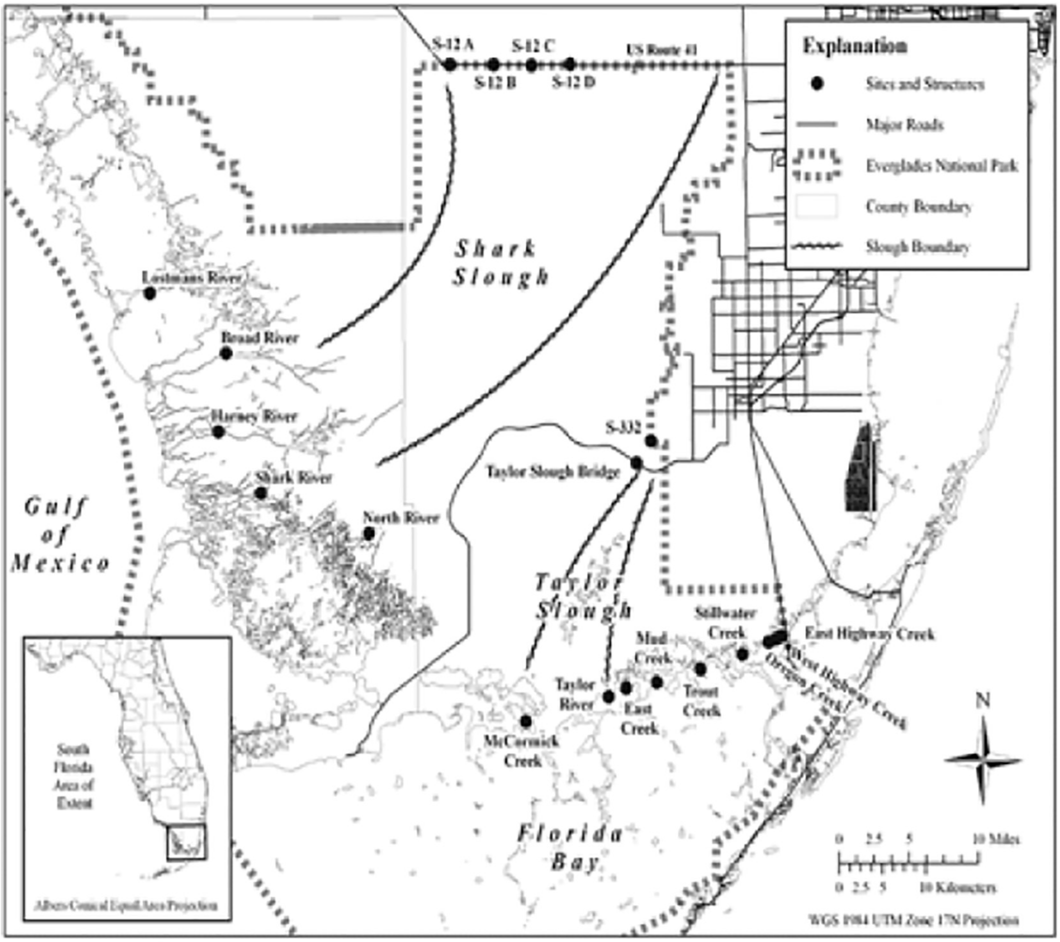

Fig. 1. The map of the Everglades National Park: Shark River boundary, the location of S12 and S333 hydrological structures and the southwest outflow tributaries. (Source: https://sofia.usgs.gov/publications/papers/swdis_salmon/images/fig1x.gif).

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. Conceptual model

Fig. 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

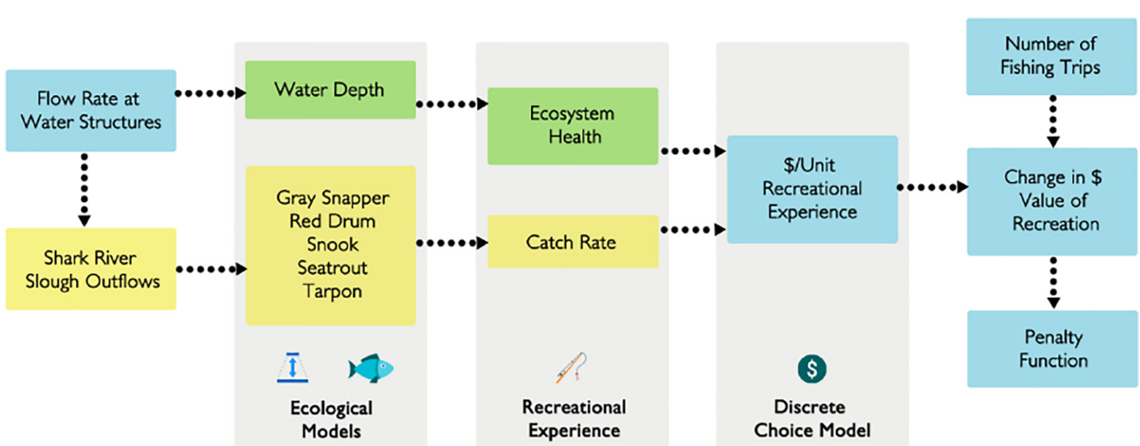

Fig. 2. Integrated framework for developing ecological-economic penalty function for managing freshwater flows in the Florida Everglades.

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 due to maintaining periodic water flows below the targets. The rest of this section explains various hydrological, ecological, and economic sub-components of the model.

2.2.1. 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 Eq. (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 Eq. (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 much more 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 this 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 6 N. These zones include Whitewater Bay, Shark River, Harney River, Broad River, Tarpon Bay, and Lostmans River.

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

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); 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 Eq. (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 due to outflows, percolation, seepage, and evapotranspiration. 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 et al. (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 due to 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 et al. (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 s^{-1} and 0.34 cm s^{-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 Eq. (2) with different lag periods, but found the one-month lag model to be the best fit.

By plugging Eqs. (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 Eqs. (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 2 × 2, as it has a 2-mile by 2-mile fixed-resolution grid system covering an area of 7600 mile^2. 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 et al., 2012), our own consultation with certain 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. (2011) 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., 2005a, b). This 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 (Eq. (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 (Eqs. (1), (2), and (4)). We resolved this 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 Eq. (4) (the depth-flow model), we had to apply the Cochrane-Orcutt transformation twice.

2.2.2. Penalty function development

The penalty in this 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 due to changes in natural factors such as rainfall, evapotranspiration, and outflows in the SRS watershed itself. Since the focus of this 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 based on the results 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.

$\Delta Y_{c,m}$ 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 Eqs. (6) and (7), substituting in Eq. (5) for ΔC_m, and simplifying the results, we obtain,

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

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,

$$P_{c,m}(F_{m-1}) = Z_{c,m}\left(F^t_{m-1}\right) - Z_{c,m}(F_{m-1}), \tag{10}$$

where $F^t_{m\,-\,1}$ is the flow target in $m-1$. Fig. 3 represents Eq. (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 $F_{m\,-\,1} > F^t_{m\,-\,1}$.

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

$$P_{c,m}(F_{m-1}) = 100 a_m w_c \left[\frac{C^t_m\left(F^t_{m-1}\right) - C_m(F_{m-1})}{C^k_m\left(F^k_{m-1}\right)}\right] \tag{11}$$

Note that $P_{c,\,m}(F_{m\,-\,1})$ is the difference between catch rates at the target flow $(F^t_{m\,-\,1})$ and the actual flow $(F_{m\,-\,1})$ for a given month, weighted by the catch rate at the current flow $(F^k_{m\,-\,1})$, 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 Eq. (3). Also, the flow-induced shortage in catch in Eq. (11), $C^t_m(F^t_{m\,-\,1}) - C_m(F_{m\,-\,1})$, is above weighted by the current catch rate $C^k_m(F^k_{m\,-\,1})$. This 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 $[P^a_{c,\,m}(F_{m\,-\,1})]$ as a weighted average of individual species catch penalties,

$$P^a_{c,m}(F_{m-1}) = \sum_{i=1}^{5} \omega_i \left\{100 a_m w_c \left[\frac{C^t_{i,m}\left(F^t_{m-1}\right) - C_{i,m}(F_{m-1})}{C^k_{i,m}\left(F^k_{m-1}\right)}\right]\right\}, \tag{12}$$

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

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

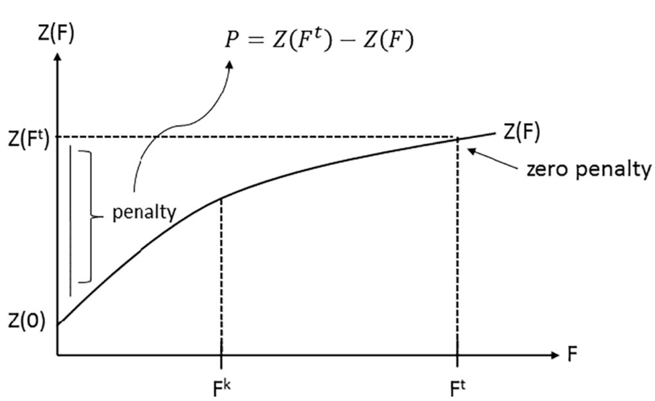

Fig. 3. Total economic recreational catch value in relation to flow.

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 Eq. (4) which connects the water depth $[D_m(F_m)]$ to water management, i.e., managed flow variable, F_m. We used this equation 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, $\frac{D^t_m(F^t_m) - D_m(F_m)}{D^k_m(F^k_m)}$. 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 this study. As the focus of this 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.,

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

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 Eqs. (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

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

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 due to 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 et al., 1994). To help ensure agents made informed decisions, a number of multimedia tools were used within the anglers' survey in this study. Two videos, each approximately 1 min 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. This 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). In order 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. This 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 is based on 600 completed surveys.

The parameters of the random utility discrete choice model in Eq. (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

Table 1
Attribute levels in choice experiment design.

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

[a] Scenario 1.
[b] Scenarios 2 and 3.

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 Eq. (12).

2.2.3. Estimation of monthly recreational Trips

The penalty function in Eq. (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 this period 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,

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

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 this problem 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) based on an aerial survey of recreational vessels and trailers in ENP waters and parking lots, respectively.

3. Results

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 Fig. 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% 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 model based CPUE at target flow levels to current conditions. The lowest estimated deficits were in the summer months (June through August). This is probably due to 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 due to decreased predation of juveniles by snook (Boucek and Rehage, 2013a, b, c; Hallac et al., 2010) or possibly due to anglers simply having switched their effort to red drum. By 2013, all species were returning to previous CPUE with a slight upward trend for snapper.

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

Table 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

[1] Baseline levels are based on estimated average historical values.

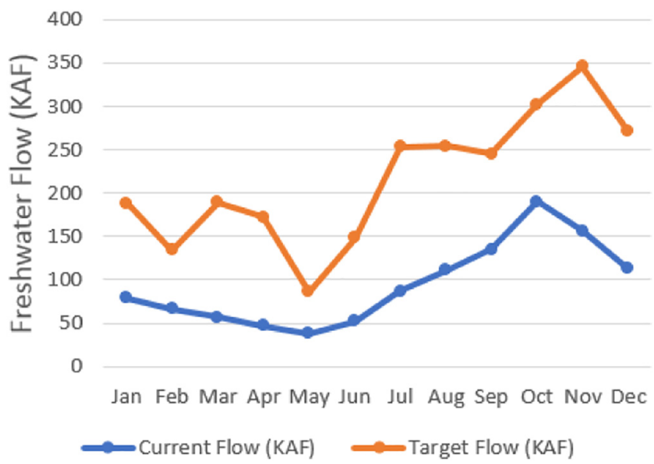

Fig. 4. Three year average current flow and target flow at the ENP node (S12 + S333 structures).

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, b; 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., 2012). By combining the results of this last model [Eq. (4)] with those of catch-flow functions [Eq. (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. This integration will allow us to analyze the effects of changes in freshwater management in SRS on fishery ecosystem system services.

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

Table 3
Estimated models of catch-flow and depth-flow relationships.

Model	Variable	Coefficient	Std error	Adjusted R^2	N	Durbin-Watson
Snook catch [Eq. (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 [Eq. (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 [Eq. (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 [Eq. (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 [Eq. (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 [Eq. (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 [Eq. (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 [Eq. (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				

* $p < .01$; ** $p < .05$; *** $p < .10$.

Table 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.
* $p < .01$.
** $p < .05$.
*** $p < .10$.

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. This is 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. MWTP 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. This price 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 based on 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% increase in annual trips over the ten-year period beginning in 2005, which saw 38,284 trips. Based on an 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.

3.4. Fisheries penalty functions

We used Eqs. (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

Table 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 $)											
0	5.38	3.82	21.22	19.35	9.53	1.99	3.38	4.00	3.08	3.60	4.13	7.64
50	3.96	2.38	15.63	13.75	4.01	1.30	2.71	3.23	2.45	3.00	3.53	6.24
100	2.53	0.94	10.04	8.14	0	0.61	2.04	2.45	1.82	2.40	2.94	4.85
150	1.11	0	4.45	2.53	0	0	1.37	1.68	1.19	1.80	2.34	3.45
200	0	0	0	0	0	0	0.71	0.91	0.55	1.20	1.75	2.06
250	0	0	0	0	0	0	0.04	0.13	0	0.60	1.16	0.66
300	0	0	0	0	0	0	0	0	0	0	0.56	0
350	0	0	0	0	0	0	0	0	0	0	0	0
Marginal value ($ AF^{-1})	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^{-1})							41.54 (11.88–112.11)					
Mean marginal value for ecosystem health only (min–max) ($ AF^{-1})							39.36 (10.05–109.05)					
Value of water in the US (Frederick et al., 1996)							In 1994 ($ AF^{-1})			In 2015 ($ AF^{-1})[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		
Value of water for agriculture (Takatsuka et al., 2018)										280.00		
Value of water for urban use (Weisskoff, 2018)										2000.00		

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

Table 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	68.81	0.00
Increase by 50% all months (scenario 1)	1132	59.67	13.23
Increase by 50% dry months (scenario 2)	766	63.37	7.92
Increase by 50% wet months (scenario 3)	1043	65.12	5.37
Increase to historical flow (scenario 4)	1040	57.98	15.75
Increase by 100% all months (scenario 5)	1509	50.52	26.58
Target level delivery (scenario 6)	2590	0.00	100.00
			Percent of baseline total
Baseline – ecosystem health only	754	64.66	93.96
Baseline – recreational fishing only	754	4.16	6.04
Baseline – recreational fishing only	754	68.81	100.00

to be high for any given level of flow, whereas during the wet months, May through October, the penalty was found to be smaller. Three factors contributed to this variation. During the wet season, the lower water shortages in relation to the target delivery kept the penalty lower. Also during those months, especially in the Fall, the total number of monthly fishing trips were lower. On the contrary, during the rest of 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 this value can be attributed to the value that anglers attach to overall ecosystem health ($39.36 AF^{-1}), while a significantly small portion to fish catch.

The implicit values of water for various uses are not readily available. Frederick et al. (1996) 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 amount 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 $2000 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.

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 $68.81 million. This total value is decomposed into two recreational attributes of fish catch at $4.16 million and overall ecosystem health at $64.66 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 $59.67 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 conducted for increasing the monthly freshwater flows to the historical levels (an annual total of 1040 KAF) and by 100% 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 (2594 KAF). The annual total penalties were reduced to $57.98 million (15.75% improvement) and $50.52 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.

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 based on improving the fishery habitat (Davis et al., 2005; Chen and Twilley, 1999; Ross et al., 2000). ENP 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 this study is probably the first attempt at linking water management variables with Everglades ecosystem services relevant to humans. Although this 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 1848 cubic feet per second (cfs) (National Park Service, 2012) to the project goal level of 4000 cfs, the penalty value of the recreational fishing experience would be lowered by 13% (based on 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. (1996) for a most comprehensive list of water prices, which are >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., 1996).

This 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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References

Abtew, W., Huebner, S., 2001. Droughts and Water Shortages in Cantral and South Florida. West Palm Beach, FL. (Retrieved from). https://my.sfwmd.gov/portal/page/portal/pg_grp_tech_pubs/portlet_tech_pubs/ema-396.pdf.

Ault, J.S., Smith, S.G., McClellan, D.B., Zurcher, N., McCrea, A., Vaughan, N.R., Bohnsack, J.A., 2008. Aerial Survey of Boater Use in Everglades National Park Marine Waters – Florida Bay and Ten Thousand Islands. NOAA Technical Memorandum NMFS-SEFSC-581 (183 pp).

Bachman, P.M., Rand, G.M., 2008. Effects of salinity on native estuarine fish species in South Florida. Ecotoxicology 17 (7):591–597. https://doi.org/10.1007/S10646-008-0244-7.

Bergstrom, J., 1990. The impact of information on environmental commodity valuation decisions. Am. J. Agric. Econ. 72 (3):614–621. https://doi.org/10.2307/1243031.

Boucek, R.E., Rehage, J.S., 2013a. Population Effects and Recovery of Common Snook (*Centropomus undecimalis*) From the 2010 Cold Front. Miami, FL.

Boucek, R.E., Rehage, J.S., 2013b. A tale of two fishes: using recreational angler records to examine the link between fish catches and floodplain connections in a subtropical coastal river. Estuar. Coasts 38 (1):124–135. https://doi.org/10.1007/s12237-013-9710-4.

Boucek, R.E., Rehage, J.S., 2013c. No free lunch: displaced marsh consumers regulate a prey subsidy to an estuarine consumer. Oikos 122 (10):1453–1464. https://doi.org/10.1111/j.1600-0706.2013.20994.x.

Brandt, L.A., Boyer, J., Browder, J., Cherkiss, M., Doren, R.F., Frederick, P., Gaiser, E., Gawlik, D., Geiger, S., Hart, K., Jeffery, B., Kelble, C., Layne, J., Lorenz, J., Madden, C., Mazzotti, F. J., Ortner, P., Parker, M., Roblee, M., Rodgers, L., Rodusky, A., Rud- nick, D., Sharfstein, B., Trexler, J., Volety, A., 2012. System-wide Indicators for Everglades Restoration. 2012 Report. Unpublished Technical Report 90 pp.

Cameron, T.A., Englin, J., 1997. Respondent experience and contingent valuation of environmental goods. J. Environ. Econ. Manag 33 (3):296–313. https://doi.org/10.1006/jeem.1997.0995.

Chen, R., Twilley, R.R, 1999. Patterns of mangrove forest structure and soil nutrient dynamics along the Shark River Estuary, Florida. Estuar. Coasts 22 (4), 955–970.

Childers, D.L., Ross, M., Leonard, L., 2005. Documenting the Importance of Water Flow to Everglades Landscape Structure and Sediment Transport in Everglades National Park. Homestead, FL Retrieved from https://www.nps.gov/ever/learn/nature/upload/MOD01-5-Final-Report.pdf.

Cochrane, D., Orcutt, G.H, 1949. Application of least squares regression to relationships containing auto-correlated error terms. J. Am. Stat. Assoc 44 (245), 32–61.

Davis, S.M., Childers, D.L., Lorenz, J.J., Wanless, H.R., Hopkins, T.E., Beach, W.P., 2005. A conceptual model of ecological interactions in the mangrove e stuaries of the Florida everglades. Wetlands 25 (4):832–842 Retrieved from https://www.researchgate.net/profile/Todd_Hopkins2/publication/226619334_A_conceptual_model_of_ecological_

interations_in_the_mangrove_estuaries_of_the_Florida_Everglades/links/568b2e2008ae1975839dc69e.pdf.
Dolan, T.J., Hermann, A.J., Bayley, S.E., Zoltek, J., 1984. Evapotranspiration of a Florida, U.S. A., freshwater wetland. J. Hydrol. 74 (3–4):355–371. https://doi.org/10.1016/0022-1694(84)90024-6.
Doren, R.F., Trexler, J.C., Gottlieb, A.D., Harwell, M.C., 2009. Ecological indicators for system-wide assessment of the greater everglades ecosystem restoration program. Ecol. Indic. 9 (6 Suppl). https://doi.org/10.1016/j.ecolind.2008.08.009.
FDHSMV, 2017. Vehicle and Vessel Reports and Statistics. Florida Department of Highway Safety and Motor Vehicles Retrieved May 1, 2017, from https://www.flhsmv.gov/resources/driver-and-vehicle-reports/vehicle-and-vessel-reports-and-statistics/.
Fedler, T., 2009. The Economic Impact of Recreational Fishing in the Everglades Region.
Frederick, K.D., VandenBerg, T., Hanson, J., 1996. Economic Values of Freshwater in the United States.pdf. Resources for the Future Discussion Paper Washington, D.C. Retrieved from http://www.rff.org/files/sharepoint/WorkImages/Download/RFF-DP-97-03.pdf.
Freeman III, M., Herriges, J.A., Kling, C.L, 1994. The measurement of environmental and resource values: theory and methods. Third edition. Washington, D.C.: Resources for the Future p. 515 pp.
Fulford, R., Yoskowitz, D., Russell, M., Dantin, D., Rogers, J., 2016. Habitat and recreational fishing opportunity in Tampa Bay: linking ecological and ecosystem services to human beneficiaries. Ecosyst. Serv. 17, 64–74.
Gunderson, L.H., 1994. Vegetation of the Everglades: determinants of community composition. In J. C. (Eds. Davis, S.M., Ogden (Ed.)). Everglades: The Ecosystem and Its Restoration. St. Lucie Press, Delray (Retrieved from). https://books.google.com/books?id=yIBAHbgWxTAC&pg=PA319&lpg=PA319&dq=Gunderson,+L.H.,+1994.+Vegetation+of+the+Everglades:+determinants+of+community+composition.+In:+Davis,+S.M.,+Ogden,+J.C.+(Eds.),+Everglades:+The+Ecosystem+and+Its+Restoration.+St.+Lucie.
Hallac, D., Kline, J., Sadle, J., Bass, S., Ziegler, T., Snow, S., 2010. Preliminary Effects of the January 2010 Cold Weather on Flora and Fauna in Everglades National Park, (January), 1–8.
Harou, J.J., Pulido-Velazquez, M., Rosenberg, D.E., Medellín-Azuara, J., Lund, J.R., Howitt, R. E., 2009. Hydro-economic models: concepts, design, applications, and future prospects. J. Hydrol. 375 (3–4):627–643. https://doi.org/10.1016/j.jhydrol.2009.06.037.
Harvey, R.G., Mazzotti, F.J., Brandt, L.A, 2011. The American Alligator: An Indicator Species for Everglades Restoration Retrieved from https://edis.ifas.ufl.edu/pdffiles/UW/UW35800.pdf.
Higman, J.B., 1967. Relationships between catch rates of sport fish and environmental conditions in Everglades National Park, Florida. Proceedings of the Gulf and Caribbean Fisheries Institute :pp. 129–140 (Retrieved from). http://aquaticcommons.org/12321/1/gcfi_19-20.pdf.
Hole, A.R., 2006. A Comparison of Approaches to Estimating Confidence Intervals for Willingness to Pay Measures. (Retrieved from). http://ideas.repec.org/p/chy/respap/8cherp.html.
Holling, C.S., Gunderson, L.H., Walters, C.J., 1994. The structure and dynamics of the Everglades system: guidelines for ecosystem restoration. In: Davis, S.M., Ogden, J.C. (Eds.), Everglades, the Ecosystem and Its Restoration. St. Lucie Press, Delray Beach, FL, pp. 741–756.
Holmes Jr., R.R., Terrio, P.J., Harris, M.A., Mills, P.C., 2001. Introduction to Field Methods for Hydrologic and Environmental Studies. (Retrieved from). https://pubs.usgs.gov/of/2001/0050/report.pdf.
Jenkins, M.W., Lund, J.R., Howitt, R.E., Draper, A.J., Msangi, S.M., Tanaka, S.K., ... Marques, G. F., 2004. Optimization of California's water supply system: results and insights. J. Water Resour. Plan. Manag. 130 (4):271–280. https://doi.org/10.1061/(ASCE)0733-9496(2004)130:4(271).
Jerath, M., Bhat, M., Rivera-Monroy, V.H., Castañeda-Moya, E., Simard, M., Twilley, R.R., 2016. The role of economic, policy, and ecological factors in estimating the value of carbon stocks in Everglades mangrove forests, South Florida, USA. Environ. Sci. Pol. 66, 160–169.
Johnston, R.J., Segerson, K., Schultz, E.T., Besedin, E.Y., Ramachandran, M., 2011. Indices of biotic integrity in stated preference valuation of aquatic ecosystem services. Ecol. Econ. 70 (11):1946–1956. https://doi.org/10.1016/j.ecolecon.2011.06.018.
Johnston, R.J., Schultz, E.T., Segerson, K., Besedin, E.Y., Ramachandran, M., 2012. Enhancing the content validity of stated preference valuation: the structure and function of ecological indicators. Land Econ. 88 (1):102–120. https://doi.org/10.1353/lde.2012.0000.
Johnston, R.J., Schultz, E.T., Segerson, K., Besedin, E.Y., Ramachandran, M., 2013. Stated preferences for intermediate versus final ecosystem services: disentangling willingness to pay for omitted outcomes. J. Agric. Resour. Econ. 42 (1), 98–118.
Kuhfeld, W.F, 2010. Experimental design: efficiency, coding, and choice designs Available from: http://support.sas.com/techsup/technote/mr2010c.pdf.
Kuhfeld, W.F., Tobias, R.D., 2005. Large factorial designs for product engineering and marketing research applications. Technometrics 47 (2), 132–141.
Lancaster, K.J, 1966. A new approach to consumer theory. J. Polit. Econ 74 (2), 132–157.
Levesque, V., 2004. Water flow and nutrient Flux from five estuarine rivers along the southwest coast of the Everglades National Park, Florida, 1997–2001. U.S. Geological Survey (Retrieved from). https://pubs.usgs.gov/sir/2004/5142/.
Lorenz, J.J., 2014. A review of the effects of altered hydrology and salinity on vertebrate fauna and their habitats in northeastern Florida bay. Wetlands 34 (Suppl. 1). https://doi.org/10.1007/s13157-013-0377-1.
Mazzotti, F.J., Harvey, R.G., Rice, K.G., Cherkiss, M.S., Jeffery, B.M, 2008. The Crocodilian Indicator in the Greater Everglades: 2006 Assessment Report. Davie, FL Retrieved from http://crocdoc.ifas.ufl.edu/publications/reports/2006_Crocodilian_Assessment_Report.pdf.
McCormick, B., Clement, R., Fischer, D., Lindsay, M., Watson, R., 2010. Measuring the Economic Benefits of America's Everglades Restoration. Retrieved from http://everglades.3cdn.net/8edd03d0943ae993fe_e0m6i4gx2.pdf.
Mirchi, A., Watkins, D.W., Engel, V., Sukop, M.C., Czajkowski, J., Bhat, M., Rehage, J., Letson, D., Takatsuka, Y., Weisskoff, R., 2018. A hydro-economic model of South Florida water resources system. Sci. Total Environ. (This issue).
Mitchell, R.C., Carson, R.T, 1989. Using surveys to value public goods: the contingent valuation method. Washington, D.C.: Resources for the Future 463 pp.
National Park Service, 2009. Potential Ecological Consequences of Climate Change in South Florida and the Everglades: 2008 Literature Synthesis (Homestead, Florida).
National Park Service, 2012. Tamiami Trail Modifications: Modified Water Deliveries and Next Steps Project. (Retrieved from). https://www.nps.gov/ever/learn/nature/upload/Hi-ResTTModsFact-SheetMay2012.pdf.
Newlin, B.D., Jenkins, M.W., Lund, J.R., Howitt, R.E., 2002. Southern California water markets: potential and limitations. J. Water Resour. Plan. Manag. 128 (1):21–32. https://doi.org/10.1061/(ASCE)0733-9496(2002)128:1(21).
Ogden, J.C., Davis, S.M., Barnes, T.K., Jacobs, K.J., Gentile, J.H., 2005a. Total system conceptual ecological model. Wetlands 25 (4):955–979. https://doi.org/10.1672/0277-5212(2005)025[0955:TSCEM]2.0.CO;2.
Ogden, J.C., Davis, S.M., Jacobs, K.J., Barnes, T., Fling, H.E., 2005b. The use of conceptual ecological models to guide ecosystem restoration in South Florida. Wetlands 25 (4): 795–809. https://doi.org/10.1672/0277-5212(2005)025[0795:TUOCEM]2.0.CO;2.
Ogden, J.C., Baldwin, J.D., Bass, O.L., Browder, J.A., Cook, M.I., Frederick, P.C., et al., 2014. Waterbirds as indicators of ecosystem health in the coastal marine habitats of southern Florida: 1. Selection and justification for a suite of indicator species. Ecol. Indic. 44:148–163. https://doi.org/10.1016/J.ECOLIND.2014.03.007.
Osborne, J., Schmidt, T.W., Kalafarski, J., 2006. Year 2005 Annual Marine Fisheries Report (Everglades National Park).
de Palma, A., Ben-Akiva, M., Brownstone, D., Holt, C., Magnac, T., McFadden, D., et al., 2008. Risk, uncertainty and discrete choice models. Mark. Lett 19 (3–4):269–285. https://doi.org/10.1007/s11002-008-9047-0.
Perry, W.B., 2008. Everglades restoration and water quality challenges in south Florida. Ecotoxicology 17 (7):569–578. https://doi.org/10.1007/s10646-008-0240-y.
Price, R.M., Swart, P.K., Willoughby, H.E., 2008. Seasonal and spatial variation in the stable isotopic composition of precipitation in south Florida. J. Hydrol. 358 (3–4):193–205. https://doi.org/10.1016/j.jhydrol.2008.06.003.
Rand, G.M., Bachman, P.M., 2008. South Florida ecosystems. Ecotoxicology 17 (7): 565–568. https://doi.org/10.1007/s10646-008-0235-8.
Richardson, C.J., 2010. The Everglades: North America's subtropical wetland. Wetl. Ecol. Manag. 18 (5):517–542. https://doi.org/10.1007/s11273-009-9156-4.
Richardson, L., Keefe, K., Huber, C., Racevskis, L., Reynolds, G., Thourot, S., Miller, I, 2014). Assessing the value of the Central Everglades Planning Project (CEPP) in Everglades restoration: an ecosystem service approach. Ecol. Econ 107, 366–377.
Ross, M., Meeder, J., Sah, J., Ruiz, P., Telesnicki, G., 2000. The southeast saline Everglades revisited: 50 years of coastal vegetation change. J. Veg. Sci. 11 (1), 101–112 Retrieved from http://www.jstor.org/stable/3236781.
Rudnick, D.T., Ortner, P.B., Browder, J.A., Davis, S.M., 2005. A conceptual ecological model of Florida Bay. Wetlands 25 (4):870–883. https://doi.org/10.1672/0277-5212(2005)025[0870:Acemof]2.0.Co;2.
Rutherford, E.S., Tilmant, J.T., Thue, E.B., Schmidt, T.W, 1989a. Fishery harvest and population dynamics of gray snapper, *Lutjanus griseus*, in Florida Bay and adjacent waters. Bull. Mar. Sci 44 (1), 139–154.
Rutherford, E.S., Tilmant, J.T., Thue, E.B., Schmidt, T.W, 1989b. Fishery harvest and population dynamics of spotted seatrout, *Cynoscion nebulosus*, in Florida Bay and adjacent waters. Bull. Mar. Sci. 44 (1), 108–125.
Saha, A.K., Moses, C.S., Price, R.M., Engel, V., Smith, T.J., Anderson, G., 2012. A hydrological budget (2002–2008) for a large subtropical wetland ecosystem indicates marine groundwater discharge accompanies diminished freshwater flow. Estuar. Coasts 35 (2):459–474. https://doi.org/10.1007/s12237-011-9454-y.
Schultz, E.T., Johnston, R.J., Segerson, K., Besedin, E.Y., 2012. Integrating ecology and economics for restoration: using ecological indicators in valuation of ecosystem services. Restor. Ecol. 20:304–310. https://doi.org/10.1111/j.1526-100X.2011.00854.x.
Sklar, F.H., Fitz, H.C., Wu, Y., Van Zee, R., McVoy, C., 2001. The design of ecological landscape models for Everglades restoration. Ecol. Econ. 37 (3):379–401. https://doi.org/10.1016/S0921-8009(01)00180-X.
Sklar, F.H., Chimney, M.J., Newman, S., McCormick, P., Gawlik, D., Miao, S., ... Rutchey, K., 2005. The ecological–societal unterpinnings of Everglades restoration. Front. Ecol. Environ. 3 (3):161–169. https://doi.org/10.1890/1540-9295(2005)003[0161:TEUOER]2.0.CO;2.
South Florida Water Management District, 2005. Documentation of the South Florida Water Management Model Version 5.5. (Retrieved from). https://www.sfwmd.gov/sites/default/files/documents/sfwmm_final_121605.pdf.
South Florida Water Management District, 2014. A Review and Evaluation of the Minimum Flow and Level Criteria for Northeastern Florida Bay. (Retrieved from). https://www.sfwmd.gov/sites/default/files/documents/nefb_mfl_update_support_document_june_2014.pdf.
Stabenau, B.E., Engel, V., Sadle, J., Pearlstine, L., 2011. Sea-level rise: observations, impacts, and proactive measures in Everglades National Park. Park. Sci. 28 (2), 26–30.
Takatsuka, Y., Niekus, M.R., Harrington, J., Watkins, D.W., Mirchi, A., Nguyen, H., Sukop, M. C., 2018. Value of irrigation water usage in south Florida agriculture. Sci. Total Environ. (This issue).
Tilmant, J.T., Rutherford, E.S., Thue, E.B, 1989. Fishery harvest and population dynamics of red drum (*Sciaenops ocellatus*) from Florida Bay and adjacent waters. Bull. Mar. Sci 44 (1), 126–138.
Train, K.E., 2009. Discrete choice methods with simulation. Cambridge University Press, Cambridge, UK.
VanZee, R., 1999. Natural System Model version 4.5 documentation report. South Florida Water Management District, West Palm Beach, FL.

Vojáček, O., Pecáková, I., 2010. Comparison of discrete choice models for economic environmental research. Prague Econ. Pap 19 (1):35–53. https://doi.org/10.18267/j.pep.363.
Walters, C., Gunderson, L., Holling, C.S., 1992. Experimental policies for water management in the Everglades. Ecol. Appl. https://doi.org/10.2307/1941775.
Wang, J.D., Luo, J., Ault, J.S, 2003. Flows, salinity, and some implications for larval transport in south Biscayne Bay, Florida. Bull. Mar. Sci 72 (3), 695–723.
Weisskoff, R., 2018. Looking back into the future: water forecasting and economic modeling for the Everglades region, A 10-year perspective. Sci. Total Environ. (This issue).
Woods, J., 1994. Surface Water Discharge and Salinity Monitoring of Coastal Estuaries in Everglades National Park, USA, in Support of the Comprehensive Everglades Restoration Plan, (954).