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Trade-offs between socioeconomic and conservation management objectives in stock enhancement of marine recreational fisheries

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ABSTRACT

We used an integrated bio-economic model to explore the nature of tradeoffs between conservation of fisheries resources and their use for socioeconomic benefit, as realized through the stock enhancement of recreational fisheries. The model explicitly accounted for the dynamics of wild, stocked, and naturally recruited hatchery-type fish population components, angler responses to stocking, and alternative functional relationships that defined conservation and socioeconomic objectives. The model was set up to represent Florida's red drum (*Sciaenops ocellatus*) fishery as a case study. Stock enhancement produced strong trade-offs characterized by frontiers indicating that maximizing socioeconomic objectives could only be achieved at great losses to conservation objectives when the latter were based exclusively on abundance of wild-type fish. When naturally recruited hatchery-type fish were considered equivalent to wild fish in conservation value, this tradeoff was alleviated. Frontier shapes were sensitive to alternative assumptions regarding how conservation objectives were formulated, differential harvesting of stocked and wild-type fish, and potential inherent stakeholder satisfaction from the act of stocking. These findings make more explicit the likely opportunity costs associated with recreational stock enhancement and highlight the utility of trade-off frontiers for evaluating management actions.

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1. Introduction

Management of recreational fisheries, like for most natural resources, is characterized by both conservation and socioeconomic objectives (Shea, 1998; Mardle and Pascoe, 2002; Walters and Martell, 2004). Conservation objectives might include valuing abundance of wild fish populations for inherent reasons (e.g., endangered species) or future benefits, including sustained harvests or yet-unrealized benefits (Cowx et al., 2010; Cooke et al., 2015). Alternatively, socioeconomic objectives commonly entail valuing a fish population for direct use—namely angler satisfaction related to catch or market activity related to fishing effort (McConnell and Sutinen, 1979; Propst and Gavrilis, 1987; Anderson, 1993). Over the long run these objectives are complimentary (Hilborn, 2007). In the short term they may conflict, since fish populations cannot be simultaneously maximally conserved and used (Sylvia and Cai, 1995; Koehn, 2010). This conflict can result in a present-time trade-off characterized by achieving short term socioeconomic objectives at the dissipation of the long term conservation objectives (Walters and Martell, 2004; Cheung and Sumaila, 2008). Selecting a satisfactory compromise between both objectives and identifying suitable management actions to realize it are thus the primary challenges of fisheries management (Walters and Martell, 2004). This challenge is acute in open access recreational fisheries. Here traditional management actions (e.g. size, bag limits) may be ineffective at controlling harvest or sustaining catch rates if captured and subsequently released fish are subject to substantial discard mortality (Coggins et al., 2007) or behavioral alterations (Camp et al., 2015). Direct control of fishing effort would be potentially effective, but is particularly unpopular with stakeholders and considered by managers to have a high socioeconomic cost (Sullivan, 2003; Dorow et al., 2010; McClenachan, 2013). To avoid these high costs while sustaining populations of fish, alternative management strategies are increasingly considered.

An alternative management strategy that restricts neither catch nor effort is stock enhancement: the release of hatchery-reared fish into waters containing wild populations of the same species (Lorenzen, 2005; Camp et al., 2014). Stock enhancement is widely used in the management of inland and increasingly, marine recreational fisheries (Richards and Rago, 1999; Halverson, 2008; Vega, 2011). The popularity of stock enhancement stems in part from the perception that this strategy can maintain or

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increase fish population abundance, catches, and fishing effort, and thereby alleviate trade-offs between conservation and socioeconomic objectives (Taylor et al., 2005). However, this should not be assumed (Lorenzen, 2014). Principally, trade-offs between conservation and socioeconomic objectives arise in stock enhancements because hatchery-reared fish may differ biologically from their wild conspecifics and may not be afforded the same conservation or utilitarian value as the latter. Biological interactions between hatchery-reared and wild fish may result in a reduction of the abundance of fish with wild characteristics, even when overall abundance of fish, catches and fishing effort are increased by the enhancement (Camp et al., 2014). To assess the nature of such tradeoffs requires considering at least three issues: (1) biological differences between wild fish, hatchery-reared fish and their naturally recruited offspring; (2) biological and fishing effort feedbacks by which stocking affects wild fish; and (3) the functional composition of socioeconomic and conservation objectives.

Hatchery rearing influences the biology of stocked fish through developmental and genetic mechanisms and often results in fish that are less fit in natural environments than their wild conspecifics and may also differ in their genetic diversity or structure (Lorenzen et al., 2012). Therefore, released stocked fish and their offspring are not, in general, fully equivalent to wild fish (Araki et al., 2008; Fraser, 2008). Once released, stocked fish (and eventually their offspring) may interact biologically with wild fish through competition, predation and reproduction (Weiss and Schmutz, 1999; Ham and Pearsons, 2001; Bell et al., 2008). Interactions may be particularly strong and immediate if stocked fish are released at small sizes because density dependent mortality is strongest in the early juvenile stages of the fish life cycle (Lorenzen, 2008; Camp et al., 2014). Exposure to density dependent processes may cause stocked and wild fish to experience increased mortality and may result in partial displacement of wild by hatchery-reared fish (Lorenzen, 2005). Differences between hatchery-reared and wild fish are at least in part genetically based and replacement may therefore persist for multiple generations, though natural selection will tend to restore wild traits and levels of fitness within a several generations (Quinn et al., 2001). Replacement of wild fish by hatchery-reared fish and their offspring may take place with or without any associated increase in total population abundance (additive or non-additive effect of stocking) (Rogers et al., 2010). If stocked fish augment overall fish populations, stocking can potentially translate into greater socioeconomic objectives via increased catch rates and related angler utility (Anderson, 1993; Schuhmann, 1998; Anderson and Lee, 2013) or increased effort and greater regional market activity (Hilborn, 1998; Camp et al., 2013). Even if enhancement does successfully augment overall fish populations in open-access fisheries, angling effort may increase in response and lead to greater fishing related mortality on wild fish (Baer and Brinker, 2010), or prevent increases in catch rates from persisting (van Poorten et al., 2011; Camp et al., 2014). Where they occur, such negative impacts on wild populations can be considered conservation costs of improving socioeconomic objectives (Camp et al., 2014), depending on the definition of those objectives (Lackey 1998; Hilborn 2007).

The capacity for any management action, such as stock enhancement, to address trade-offs between socioeconomic and conservation objectives ultimately depends on the characteristics of those objectives (Lackey, 2004; Koehn, 2010). Objectives can be functionally characterized by the relationships between objective value and changes in measurable outcomes, such as catch rate or wild fish abundance (Hilborn, 2007; Koehn and Todd, 2012). These functional relationships can be strongly influenced by societal perceptions and preferences (Lackey, 2003; Arlinghaus, 2005). For example, the value of socioeconomic objectives achieved via enhancement depends on the functional relationship between catch-related satisfaction and marginal increase in catch rates (Camp et al., 2013), as well as the strength of any inherent stakeholder preferences for or against stocking as a management action (Baer and Brinker, 2010; Arlinghaus et al., 2014). Similarly, the conservation value associated with stock enhancement largely depends on how society views wild versus stocked fish (Myers et al., 2004; Olaussen and Liu, 2011; Anderson and Lee, 2013), but also on the marginal values of each unit of wild fish (Cooke et al., 2015)—i.e. the value of one unit of wild stock over a range of stock sizes, from unfished conditions to extinction. While some societal preferences have been well studied, such as marginal increases in angler satisfaction from additional catches (Arlinghaus et al., 2014; Beardmore et al., 2015), others have not been, such as inherent preferences for management strategies or marginal value of wild fish (Holling and Meffe, 1996; van Poorten et al., 2011; Arlinghaus et al., 2014). In fact, fundamental goals, clear objectives and explicit, quantitative targets are often not well defined for the socioeconomic and conservation management of many recreational fisheries (Lackey, 1998; Walters and Martell, 2004; Cooke et al., 2015). This creates a real challenge for assessing trade-offs between objectives realized under certain management strategies, like stock enhancement.

One approach to assess how potential management actions address trade-offs between even implicit objectives involves using trade-off frontiers to visualize the relative socioeconomic and conservation opportunity costs—that is, what is sacrificed from one objective to achieve some amount of the other (Possingham and Shea, 1999; Walters and Martell, 2004; McNie, 2007). Relative opportunity costs can be characterized for a given strategy (e.g., stock enhancement) by assessing the conservation and socioeconomic outcomes realized under a range of implementations (e.g., number and size of fish stocked). Plotting these anticipated outcomes against each other on a plane visualizes the conservation and socioeconomic outcomes possible with specific implementations of the given strategy (Walters and Martell, 2004). The Pareto-efficient implementation options comprising the outermost points represent the "frontier" for a strategy (Sylvia and Enriquez, 1994; Cheung and Sumaila, 2008; Lester et al., 2013 Cheung and Sumaila, 2008; Lester et al., 2013). The frontier shape reveals something of the nature of the trade-off and has management implications (Walters and Martell, 2004; Cheung and Sumaila, 2008). A concave down shape suggests an opportunity cost-efficient compromise is possible (e.g., a certain size and number of fish stocked provides high conservation and socioeconomic outcomes relative to alternative stocking implementation). Alternatively, a concave up shape would suggest high opportunity costs of a compromise, such that the most efficient implementations would focus on achieving only one objective (Walters and Martell, 2004). While assessment of trade-off frontiers is not uncommon (e.g., Figge, 2004; Sanchirico et al., 2008; Gaydon et al., 2012), it has rarely been completed for recreational fisheries management strategies, and to our knowledge never with stock enhancement.

The overall objective of this work was to explore how stock enhancement might be expected to address conservation-socioeconomic trade-offs common to recreational fisheries. Specifically, we evaluated the nature of the trade-off frontiers realized with stock enhancement, and how these frontier shapes might be sensitive to alternative assumptions regarding the composition of objectives and the possible use of differential harvesting of stocked and wild fish. To accomplish this we used an integrated bioeconomic model to systematically assess socioeconomic and conservation outcomes of alternative implementations of stock enhancement, and used these outcomes to depict stylized trade-off frontiers. The results provide insights into the efficacy of stocking programs to simultaneously achieve economic value while maintaining conservation value associated with healthy wild stocks.

2. Materials and methods

2.1. Case study

We used a red drum (*Sciaenops ocellatus*) recreational fishery as a case study for this work. Red drum are a long lived (40 year lifespan) and large (adults >1 m length, 20 kg weight) species common in the Gulf of Mexico and southern North Atlantic. The red drum fishery is predominantly recreational and focused on sub-adult fish (ages 1–4 years) that remain in estuaries and inshore waters and are easily accessed by anglers. Stock enhancement programs for red drum exist on a large scale in Texas (Gulf of Mexico) and have been carried out experimentally in Florida (Gulf) and in South Carolina (Atlantic) (Tringali et al., 2008; Vega, 2011; Denson et al., 2012). This work considers potential red drum stock enhancement of a popular Florida estuary, Tampa Bay, where a hatchery is currently planned.

The marine recreational fishery is considered important to Florida's recreational fishing economy, as red drum are one of the most popular targets for anglers due to its perceived attributes as a sportfish, year-round availability and widespread distribution. Florida manages the recreational red drum fishery using size and bag limits, with a population target equivalent to 40% that of unfished wild spawning biomass, and the most recent stock assessments do not consider red drum to be overfished or undergoing overfishing (Murphy and Muyandorero, 2009). However, increasing fishing effort has led to concerns of overfishing in the near future (Murphy and Muyandorero, 2009), and this concern motivates interest in stock enhancement as a means to augment or sustain catch rates and market activity related to fishing trips without compromising the fish populations (Camp et al., 2013).

2.2. Biological sub-model

To assess the trade-offs of stock enhancement, we developed an integrative, bioeconomic simulation model. This model follows a structure similar to that described in detail in Camp et al. (2014), but has been extended to consider broad socioeconomic and conservation objectives and alternative assumptions of their composition. Model variables and parameters are summarized in Table 1. Table 2 summarizes the modeling equations. Fish populations were represented using annual, discrete-time, age-structured accounting (Table 2, Equation E.1–E.6 and E.30–E.36). Following Lorenzen (2005), the model tracked the abundances of three population components—wild (N_{a,t_w}), stocked (N_{a,t_s}), which are directly released from hatcheries and hatchery fish (N_{a,t_h}), defined as the wild-born progeny of stocked fish (Table 2, E.32). Total population egg production is divided into wild-type and hatchery-type eggs in proportion to the parental contributions (assuming that both, stocked and naturally recruited hatchery fish produce hatchery-type offspring). Natural selection will act to move the average performance of the combined population towards that of the wild component, and this process is modelled as transition of eggs from the hatchery component into the wild component at a rate equal to the heritability parameter (γ_{h}).

Both the size and density-dependent components of mortality each sub-population experienced during the first year of life were represented by "unpacking" recruitment dynamics (Table 2, E.6–E.24) following the methodology described by Lorenzen (2005). This approach subjected fish stocked at smaller sizes to cumulatively more density dependent mortality than those stocked closer to the size of "recruitment"—here defined as the cessation from density-dependent mortality. This accounted for the possibility that stocked fish may replace some would-be wild recruits as a result of competition during the density-dependent mortality life history stage.

2.3. Fishery sub-model

With no commercial harvest in Florida, the model assumed fishing mortality was due solely from recreational angling. Dome shaped vulnerability-at-age to recreational capture (v_a^{c}) was assumed since small fish are invulnerable to angling gear and larger fish move off-shore where they are rarely encountered by anglers (Table 2, E.25). Vulnerability-at-age to recreational harvest (v_a^{h}) was also assumed dome shaped (Table 2, E.27) given the length-based harvest window management strategy implemented in Florida which allows only harvest of red drum greater than 18″ and less than 27″. The model accounted for discard mortality (Table 1, D), both of captured fish greater or lesser than the harvestable size ($S_{t\mathrm{d}}$), or those within the harvest window that were voluntarily released ($S_{t\mathrm{k}}$) (Table 2, E.35 and E.36). To explore differential harvesting of stocked or wild fish, we altered the k parameter (Table 1) according to different population components (e.g. a separate k for wild and hatchery versus stocked fish), which are used in E.34 & E.36 (Table 2). We first assumed that only wild fish would be harvested, and so set $k = 0$ for stocked fish, and then assumed the inverse—that $k = 0$ for wild and hatchery fish. For cases where k was not assumed to be 0, it was assumed to be 0.27 as estimated—i.e. that substantial voluntary release of harvestable fish occurred. The number of fishing trips (aggregate effort,F_t) was a critical component of this model, because it linked the biological sub-model (where effort is a factor of fishing harvest rate, U_t, Table 2, E. 31) to the socioeconomic sub-model, where effort is a factor of total socioeconomic value (described below). This accounted for responses of effort to changes in vulnerable fish populations, such as might occur with stock enhancement. Aggregate fishing effort in each year was related to the total number of vulnerable fish in the previous year via a logistic relationship (Table 2, E.29), as is common in fisheries models (Walters and Martell 2004; Allen et al., 2013; Camp et al., 2014), and as shown empirically for Florida Gulf red drum fisheries (Camp et al., 2016).

2.4. Socioeconomic sub-model

The socioeconomic sub-model used for this work represented the total socioeconomic value to anglers in year t (V_t, Table 2, E. 41) generated from the fishery by scaling the satisfaction per trip ($A_{t_{\mathrm{t}}}$, Table 2, E. 40) by the annual total number of trips (F_t). Recreational fisheries are often valued using per-trip metrics (Schuhmann 1998), and here satisfaction per trip in year t was calculated as the sum of two components; weighted catch-rate-related satisfaction ($\delta A_{t_{\mathrm{c}}}$, Table 2, E.39 and 40) per Cox, Walters and Post (2003), and non-catch-rate-related satisfaction ($A_{t_{\mathrm{n}}}$, Table 1), (Hunt 2005). The representation of catch-rate-related satisfaction ($\delta A_{t_{\mathrm{c}}}$, Table 2, E.39) allowed for flexible assumptions regarding the functional response of angler satisfaction to catch rate. For example, value of $\psi = 1$ would lead to a proportional increase in catch-related satisfaction with increasing catch rate, whereas $\psi < 1$ would result in asymptotic and marginally decreasing catch-rate-related satisfaction with increasing catch rate, and $\psi > 1$ would represent catch-related satisfaction increasing exponentially with marginal increases in catch rate. Overall, this representation of socioeconomic value allowed calculation in units of satisfaction, flexibility in the functional response of angler catch-related satisfaction, a scaling by the number of trips in the fishery and has a precedent in fisheries studies aimed at providing management advice (Cox and Walters 2002; Cox et al., 2003; Camp et al., 2015).

Table 1
Description of parameters and parameter values (* indicates estimated).

Symbol	Description	Units	Value
$\bar{R}_0$	Recruitment at unfished conditions	fish	450,371
L_∞	Asymptotic length	mm	934
K	Von Bertalanffy metabolic parameter	yr^{-1}	0.46
t_0	Age at length = 0	yr^{-1}	−0.26
w_a	Weight-length constant	g	0.000000617
w_b	Weight-length exponent	g	3.09
W_m	Weight at maturity	kg	10.084
M	Instantaneous mortality at L_m	yr^{-1}	0.113
L_m	Reference length for mortality	mm	730
C_l	Allometric exponent of length-mortality relationship	constant	0.9
A_m	Maximum age	years	40
Ω	Recruitment compensation parameter	ratio	11
L_0	Length at entering recruitment period	mm	20
L_s	Length at stocking	mm	25–175
L_r	Length at leaving recruitment period	mm	180
d_1	Duration of density dependent mortality recruitment phase, from size *L0* to *Lr*	years	0.75
d_2	Duration of the second stage of the density dependent mortality recruitment stage	Proportion	Calculated
M_1	Natural instantaneous mortality $year^{-1}$ of a 10 mm fish	year	15
S_r	Cumulative base survival for the recruitment period	rate	Calculated
ρ_h	Fitness (or survival) of hatchery relative to wild, stage 1	rate	1.0
ρ_s	Fitness (survival) of stocked relative to wild, stage 2	rate	0.8
γ_h	Share of hatchery eggs inheriting wild characteristics	%	0.2
T_t	Number of fish stocked each year	fish	$0 - 10 * \bar{R}_0$
$S_{0.5}$	Back-scaled mortality to fish size midway between 0.75 yrs and 1 yrs	yr^{-1}	0.86
L_l^c, L_u^c	Fish length for vulnerability to capture: low, high	mm	400, 850
σ_l^c, σ_u^c			$0.1 * L_l^c$,$0.1 * L_u^c$
L_l^h, L_u^h	Fish length for vulnerability to harvest: low, high	mm	457, 686
σ_l^h, σ_u^h			$0.01 * L_l^h$,$0.01 * L_u^h$
$^*k$	Proportion of harvestable fish killed	%	0.27
σ	Standard deviation of logistic describing angler effort dynamics	constant	1.0
F_m, F_o	Minimum effort and effort at unfished stock size	trips	200,000; 600,000
$^*q$	Catchability coefficient	rate	0.0000051
D	Discard mortality	rate	0.08
A_c	Satisfaction from catch	rate	Calculated
A_n	Non catch-related satisfaction	constant	6.5
A_s	Stocking-related satisfaction	rate	Calculated
ψ	Angler catch-related satisfaction response to catch rate	constant	variable
δ	Ratio of catch to non-catch-related satisfaction	ratio	0–1
y_0	Catch rate prior to stock enhancement	Constant	1.23
φ	Scalar for catch related satisfaction (A_c)	Constant	3.95
χ	Logical switch for satisfaction inherent to stocking	Binomial	1 or 0
τ	Stocking-related satisfaction at low stocking	constant	variable
ξ	Shape parameter for how satisfaction increases with stocking	Constant	0.08

2.5. Calibration

The base model was calibrated and tuned to represent the red drum recreational fishery of Tampa Bay, Florida, following the methodology described in Camp et al. (2014). Briefly, the model makes use of biological and fishery parameters estimated in recent stock assessments. Scaling parameters not known at the spatial scale of Tampa Bay (recruitment at unfished conditions, R_0, catchability, q, and proportion of captured, legally harvestable fish that were actually harvested, k, Table 1), were estimated by minimizing a negative log likelihood summing log deviances between observed and model-predicted effort, catch, and harvest, exploitation rate and escapement rate for the counties surrounding Tampa Bay (NMFS, 2013). This tuning produced potentially more realistic un-stocked baseline conditions enabling more straightforward assessment of the simulated effects of stocking.

2.6. Analyses

All analyses were carried to equilibrium conditions, assuming continued annual stocking. Evaluating trade-off frontiers between socioeconomic and conservation objectives required metrics representing each objective. We defined the conservation objective (CO) to be a function of the proportion of wild spawning biomass (B_t, Table 2, E.30) and the socioeconomic objective to be represented by socioeconomic value (V_t, Table 2, E.41). The absolute conservation objective (in units B_t) and socioeconomic objectives (in units V_t) were calculated at equilibrium for each stocking implementation (e.g., size and number of fish stocked), resulting in a set of socioeconomic (V^*) and conservation (B^*) outcomes that might be simultaneously achieved. The equilibrium conservation and socioeconomic objectives (B^* and V^*) were then converted to relative values, that is, the ratio was taken of a given implementation (e.g., a specific size and number stocked) to the maximum objective value returned from the investigated alternatives. Finally, we plotted the relative conservation values against the relative socioeconomic values to visualize the trade-off frontier, per Walters and Martell (2004). To consider how trade-offs and associated frontiers were sensitive to various assumptions, analyses were categorized into three groups, described below.

2.6.1. Conservation objectives based on truly wild vs. naturally recruited fish

We first evaluated the trade-off frontier shapes under initial assumptions about the biological and socioeconomic sub-models. We assumed that angler effort (F_t, Table 2 E.29) was moderately responsive to vulnerable fish abundance ($\sigma = 1$), such that an increase in vulnerable red drum produced a proportional increase in fishing effort. We also assumed catch-related satisfaction (A_{t_c}, Table 2, E.39) was proportional to catch rate (i.e. $\psi = 1$). The conser-

Table 2
Description of model components and equations.

Eq	Component	Equation
	Life History Characteristics of Stock:	
E.1	Length (mm) L at age $_a$	$L_a = L_\infty\,(1 - \mathrm{e}^{-K(a-t_0)})$
E.2	Mass (kg) W at age $_a$	$W_a = w_a {L_a}^{w_\mathrm{b}}$
E.3	Fecundity f at age $_a$	$f_a = \max(0, (W_a - W_\mathrm{m}))$
E.4	Survival (year^{-1}) S at age $_a$	$S_a = \mathrm{e}^{(-(ML_\mathrm{m}{L_a}^{-1}))^{C_1}}$
E.5	Survivorship l at age $_a$	$l_a = 1 \quad l_a = l_{a-1}S_{a-1} \mid \dfrac{a=1}{a=2:Amax}$
E.6	Eggs per recruit ϕ_e	$\varphi_\mathrm{e} = \sum_a f_a l_a$
	Unpacked Recruitment Dynamics:	
E.7	Beverton-Holt a, b, and re-parameterized b_n	$\mathrm{a} = \Omega/\varphi_\mathrm{e}, \quad \mathrm{b} = \left(\Omega - 1\right)(R_0\varphi_\mathrm{e})^{-1}, \quad b_\mathrm{n} = \mathrm{a}b^{-1}$,
E.8	Duration of phase 1 of recruitment stage 2, d_1	$d_1 = (L_\mathrm{s} - L_0)(L_\mathrm{r} - L_0)^{-1}$
E.9	Linear growth rate (year^{-1}) for recruitment stage 2, V	$V = (L_\mathrm{r} - L_0)\,{d_1}^{-1}$
E.10	Base survival of phase 1 and 2, respectively of recruitment stage 2, S_1, S_2	$S_1 = (L_0{L_\mathrm{s}}^{-1})^{M_1V^{-1}}, S_2 = (L_\mathrm{s}{L_\mathrm{r}}^{-1})^{M_1V^{-1}}$
E.11	Survival rate of larvae for entire recruitment, ω	$\omega = a{S_\mathrm{r}}^{-1} \mid S_\mathrm{r} = S_1S_2$
E.12	Survival a_1 for phase 1 of recruitment stage 2, modified by relative fitness of wild ($_\mathrm{w}$) and hatchery ($_\mathrm{h}$) sub-populations	$a_{1_w} = S_1, \quad a_{1_\mathrm{h}} = S_1\rho_\mathrm{h}$
E.13	Survival a_2 for phase 2 of recruitment stage 2, modified by relative fitness of wild ($_\mathrm{w}$) and hatchery ($_\mathrm{h}$) and stocked ($_\mathrm{s}$) sub-populations	$a_{2_w} = S_2, \quad a_{2_\mathrm{h}} = S_2\rho_\mathrm{h}, \quad a_{2_\mathrm{s}} = S_2\rho_\mathrm{s}$
E.14	Density dependent component of survival for phases 1 and 2, respectively, of recruitment stage 2, b_1 and b_2	$b_1 = d_1\mathrm{a}{b_\mathrm{n}}^{-1}, \quad b_2 = \mathrm{a}(b_\mathrm{n} - b_1)^{-1}(S_1\omega)^{-1}$
E.15	Total eggs E_t in the beginning year $_t$, the sum of wild ($\mathrm{E_w}$) and hatchery eggs ($\mathrm{E_h}$)	$E_{t-1_\mathrm{h}} = \sum_a f_a N_{t_\mathrm{h}}\ E_{t_\mathrm{t}} = E_{t-1_\mathrm{w}} + E_{t-1_\mathrm{h}}$ where $E_{t-1_\mathrm{w}} = \sum_a f_a N_{t_\mathrm{w}}, \quad E_{t-1_\mathrm{h}} = \sum_a f_a N_{t_\mathrm{h}}$
	Number of fish N^1 surviving recruitment stage 1 and phase 1 of recruitment stage 2 in year $_t$:	
E.16	wild ($_\mathrm{w}$)	$N^1{}_{t_\mathrm{w}} = (E_{t-1_\mathrm{w}} + \gamma_\mathrm{h}E_{t-1_\mathrm{h}})\omega a_1(1 + b1E_{t_t})^{-1}$
E.17	hatchery ($_\mathrm{h}$)	$N^1{}_{t_h} = E_{t-1_\mathrm{h}}(1 - \gamma_h)\omega a_{1_\mathrm{h}}(1 + b_1E_{t_t})^{-1}$
E.18	Total fish N^2 entering phase 2 of recruitment stage 2 in year $_t$:	$N^2{}_{t_\mathrm{t}} = N^1{}_{t_\mathrm{w}} + N^1{}_{t_\mathrm{h}} + T_t$
	Number of fish R surviving phase 2 of phase 2 and thus leaving recruitment period in year $_t$:	
E.19	Wild ($_\mathrm{w}$)	$R_{t_\mathrm{w}} = N^1{}_{t_\mathrm{w}}a_{2_w}(1 + bN^2{}_{t_\mathrm{t}})^{-1}$
E.20	Hatchery ($_\mathrm{h}$)	$R_{t_\mathrm{h}} = N^1{}_{t_\mathrm{h}}a_{2_\mathrm{h}}(1 + bN^2{}_{t_\mathrm{t}})^{-1}$
E.21	Stocked ($_\mathrm{s}$)	$R_{t_\mathrm{s}} = T_ta_{2_\mathrm{s}}(1 + bN^2{}_{t_\mathrm{t}})^{-1}$
	Number N fish entering age 1, modified by back-calculated survival from 0.75 to 1 yr of age $S_{0.5}$ in year $_t$:	
E.22	Wild ($_\mathrm{w}$)at age $_a$	$N_{a=1,t_\mathrm{w}} = R_{t_\mathrm{w}}S_{0.5}$
E.23	Hatchery ($_\mathrm{h}$) at age $_a$	$N_{a=1,t_\mathrm{h}} = R_{t_\mathrm{h}}S_{0.5}$
E.24	Stocked ($_\mathrm{s}$) at age $_a$	$N_{a=1,t_\mathrm{s}} = R_{t_\mathrm{s}}S_{0.5}$
	Fishery Characteristics:	
E.25	Vulnerability v at age $_a$ to capture ($^\mathrm{c}$)	$v_a^\mathrm{c} = g_a^\mathrm{c} - p_a^\mathrm{c} \mid$ where
	upper − lower	$\mathrm{g_a^c} = (1 + \mathrm{e}^{-(\mathrm{L_a} - \mathrm{L_u^c})\sigma_\mathrm{u}^{\mathrm{c}^{-1}}})^{-1}, \quad \mathrm{p_a^c} = (1 + \mathrm{e}^{-(\mathrm{L_a} - \mathrm{L_1^c})\sigma_1^{\mathrm{c}^{-1}}})^{-1}$
E.27	Vulnerability v at age $_a$ to harvest ($^\mathrm{h}$)	$v_a^\mathrm{h} = g_a^\mathrm{h} - p_a^\mathrm{h} \mid$ where
	upper − lower	$\mathrm{g_a^h} = (1 + \mathrm{e}^{-(\mathrm{L_a} - \mathrm{L_u^h})\sigma_\mathrm{u}^{\mathrm{h}^{-1}}})^{-1}, \quad \mathrm{p_a^h} = (1 + \mathrm{e}^{-(\mathrm{L_a} - \mathrm{L_1^h})\sigma_1^{\mathrm{h}^{-1}}})^{-1}$
E.29	Effort F in year $_t$	$F_t = \dfrac{F_\mathrm{m} + 2(F_0 - F_\mathrm{m})}{\left(1 + \dfrac{\mathrm{e}^{-((\sum_a N_{t-1_\mathrm{w}}v_a^\mathrm{c} + \sum_a N_{t-1_\mathrm{s}}v_a^\mathrm{c} + \sum_a N_{t-1_\mathrm{h}}v_a^\mathrm{c}) - \sum_a N_{t=1_\mathrm{h}}v_a^\mathrm{c})}}{\sigma\sum_a N_{t=1_\mathrm{w}}v_a^\mathrm{c})}\right)}$
	Time Dynamics:	
E.30a	Wild spawning biomass B in year $_t$	$B_t = E_{t_\mathrm{w}}{E_{t=1_\mathrm{w}}}^{-1}$
E.30b	Naturally recruited spawning biomass B in year $_t$	$B_t = (E_{t_\mathrm{w}} + E_{t_\mathrm{h}})E_{t=1_\mathrm{w}}{}^{-1}$
E.31	Exploitation rate U in year $_t$	$U_t = 1 - \mathrm{e}^{qF_t}$
E.32	Numbers N at age $_a$ and year $_t$. Calculations same for wild ($_\mathrm{w}$), stocked ($_\mathrm{s}$), and hatchery ($_\mathrm{h}$) components	$N_{a,t} = N_{a-1,t-1}S_{a-1}S_{t\mathrm{v}}S_{t\mathrm{d}}S_{t\mathrm{k}}$
E.33	Survival S of all types from in year $_t$:	
E.34	Harvest ($_\mathrm{v}$)	$S_{t\mathrm{v}} = 1 - v_{a-1}^\mathrm{h}U_{t-1}$
E.35	Discard of non-legal catch ($_\mathrm{d}$)	$S_{t\mathrm{d}} = (v_{a-1}^\mathrm{c}U_{t-1} - v_{a-1}^\mathrm{h}U_{t-1})D$
E.36	Voluntary discard of legal catch ($_\mathrm{k}$)	$S_{t\mathrm{k}} = (v_{a-1}^\mathrm{h}U_{t-1}k^{-1} - v_{a-1}^\mathrm{h}U_{t-1})D$
	Socio-economic Dynamics:	
E.37	Total catch C_t in year $_t$	$C_{t_\mathrm{t}} = U_t * \left[N_{t_\mathrm{w}}(v_a^\mathrm{c}) + N_{t_\mathrm{h}}(v_a^\mathrm{c}) + N_{t_\mathrm{s}}(v_a^\mathrm{c})\right]$
E.38	Catch per unit effort y in year $_t$	$y_t = C_{t_\mathrm{t}}{F_t}^{-1}$
E.39	Catch-rate-related satisfaction	$A_{t_\mathrm{c}} = \varphi{y_t}^{\psi}{y_0}^{-1}$
E.40	Total satisfaction per trip	$A_{t_\mathrm{t}} = \delta A_{t_\mathrm{c}} + A_{t_\mathrm{n}}$
E.41	Socioeconomic value	$V_t = A_{t_\mathrm{t}}F_t$

vation objective was calculated directly from the wild spawning biomass B^* (Table 2, E.30). This assumption implied that all marginal changes in B^* would have the same marginal effect on the value of the conservation objective value (CO)—such that, for example, a decrease of B^* from 0.55 to 0.45 would represent the same decrease in CO as a decrease in B^* from 0.30 to 0.20. We also explored the implications of considering that all hatchery fish—the naturally recruited and wild-born progeny of stocked fish—were fully equivalent to wild fish with respect to contribution to the conservation objectives. Critically, this does not necessarily assume that wild and hatchery fish function identically ecologically, nor that they have identical genotype, but rather that all naturally recruited fish are afforded the same value from a conservation perspective. This assumption may be particularly reasonable if hatchery fish appear to anglers to be physically identical to their wild counterparts. This assumption was made by altering E.30 (Table 2).

2.6.2. Alternative objective functions

In the second phase of the analysis we explored how trade-off frontiers would be affected by alternative assumptions of three functional relationships composing socioeconomic and conservation objectives. For each functional relationship considered, we calculated the relative values of the socioeconomic and conservation objectives separately. This facilitated assessing the effect on trade-off frontier shapes by alternative assumptions of the specific functional relationship, but did not permit direct comparison of absolute conservation or socioeconomic objectives across the three functional relationships investigated. To ensure comparisons among all assumptions of a given functional relationship were meaningful, alternative functional forms were parameterized to return the same absolute conservation and socioeconomic values under unstocked, baseline conditions.

We first considered the trade-off frontiers under alternative assumptions regarding the functional relationship between angler catch-related satisfaction (A_{t_c}) and nominal catch rates (Table 2, E.39). This relationship may vary depending on target species (Beardmore et al., 2015) and has not been well assessed for red drum recreational fisheries (Camp et al., 2013). We compared the initial assumption of linearly increasing satisfaction to greater catch rates (represented by $\psi = 1$) to a saturating relationship ($\psi = 0.5$) that represented diminishing marginal returns to catch-related satisfaction in response to increasing catch rates, and also to an exponential relationship ($\psi = 1.5$) that implied anglers were disproportionately positively satisfied with greater catch rates. We also considered the effect of alternative assumptions of the functional relationship between CO and equilibrium wild spawning biomass (B^*), which has particular significance for the conservation value associated with marginal changes in wild populations at low abundance. The initial analysis assumed CO was equal to B^*, but here we assumed an alternative, saturating function using a logistic formulation:

$$CO = 0.623/1 + e^{-\frac{B^* - B_R}{B^* \hat{\sigma}}} \tag{1}$$

where $\hat{\sigma}$ was the standard deviation of the logistic relationship (Table 1), B_R refers to an inflection point in the logistic (Table 1) and the constant 0.623 was selected to allow conservation value realized in unstocked conditions to be consistent with that realized under the initial assumptions. The inclusion of B^* in the denominator of the exponent produces the saturating function. This saturating shape (and the formulation of Equation (1)) was selected to represent the scenario in which the conservation objective value (CO) would fall sharply only at very low proportions of remaining wild spawning stock (i.e. < 10%). As a second alternative, we considered a different parameterization of the relationship between the CO and B^*:

$$CO = 0.323/1 + e^{-\frac{B^* - B_R}{B_R \hat{\sigma}}} \tag{2}$$

where the constant (0.323) and $\hat{\sigma}$ (Table 1) take alternative values to achieve conservation value under unstocked conditions comparable to initial assumptions, and the denominator of the exponent is formulated to represent a sigmoidal decline. This implies that there would be little decrease in CO at very high or very low B^* levels, but that the conservation objective value (CO) would fall sharply as wild spawning biomass declines below a threshold of 0.35. Thresholds for most exploited species are commonly considered 0.2–0.4 (Walters and Martell 2004), and Florida has used thresholds of 0.3 and 0.4 for red drum (Murphy and Muyandorero, 2009).

2.6.3. Differential harvesting

The conservation and socio-economic implications of stock enhancement may be modified by differential harvesting of stocked and wild fish. For example, some fisheries only permit the harvest of stocked fish; wild fish must be immediately released. We explored this regulation, as well as its counterpart—that only wild fish would be harvested. The latter is most likely not enacted as a regulation, but may well be considered as social norm in situations where anglers may perceive wild fish to be preferable for consumption.

2.6.4. Inherent satisfaction from stocking

Finally we considered how inherent satisfaction from stock enhancement alone (i.e. independent of stocking impacts on catch rates) could alter the trade-off shapes. Assuming some inherent satisfaction could occur from stocking was theorized as:

$$A_{t_s} = \chi(\tau Q_t^{\xi}) \tag{3}$$

with χ representing a logical "switch" (i.e., taking the value of 1 or 0) for whether inherent satisfaction from stocking occurs, τ representing the stocking-related satisfaction at low stocking, Q is the number of fish stocked each year and ξ the shape parameter for how satisfaction increases with additional stocking. This hypothetical function represents the possibility that anglers might be satisfied with stocking regardless of impact to the overall fish population (Scharf 2000; Arlinghaus et al., 2014), such that satisfaction initially increased dramatically with any number and size of fish stocked, but then asymptotes with greater numbers of fish stocked per year. To evaluate this possibility, inherent satisfaction from stocking (Eq. (3)) was then included in the calculation of total satisfaction (Table 2, E.40), such that total satisfaction was calculated as:

$$A_{t_t} = \delta A_{t_c} + A_{t_n} + A_{t_s} \tag{4}$$

3. Results

3.1. Conservation objectives based on truly wild vs. naturally recruited fish

As modelled, an inverse trade-off between conservation and socioeconomic metrics was realized with stock enhancement when conservation objectives were based solely on wild-type fish (Fig. 1a). Conservation objectives were best preserved by not stocking at all, whereas socioeconomic objectives were best achieved by stocking great numbers of large fish. The stocking scenario outcomes translate into trade-off frontiers of generally concave-up shapes, suggesting steep increases in forgone conservation objectives (i.e. conservation opportunity costs) associated with initial

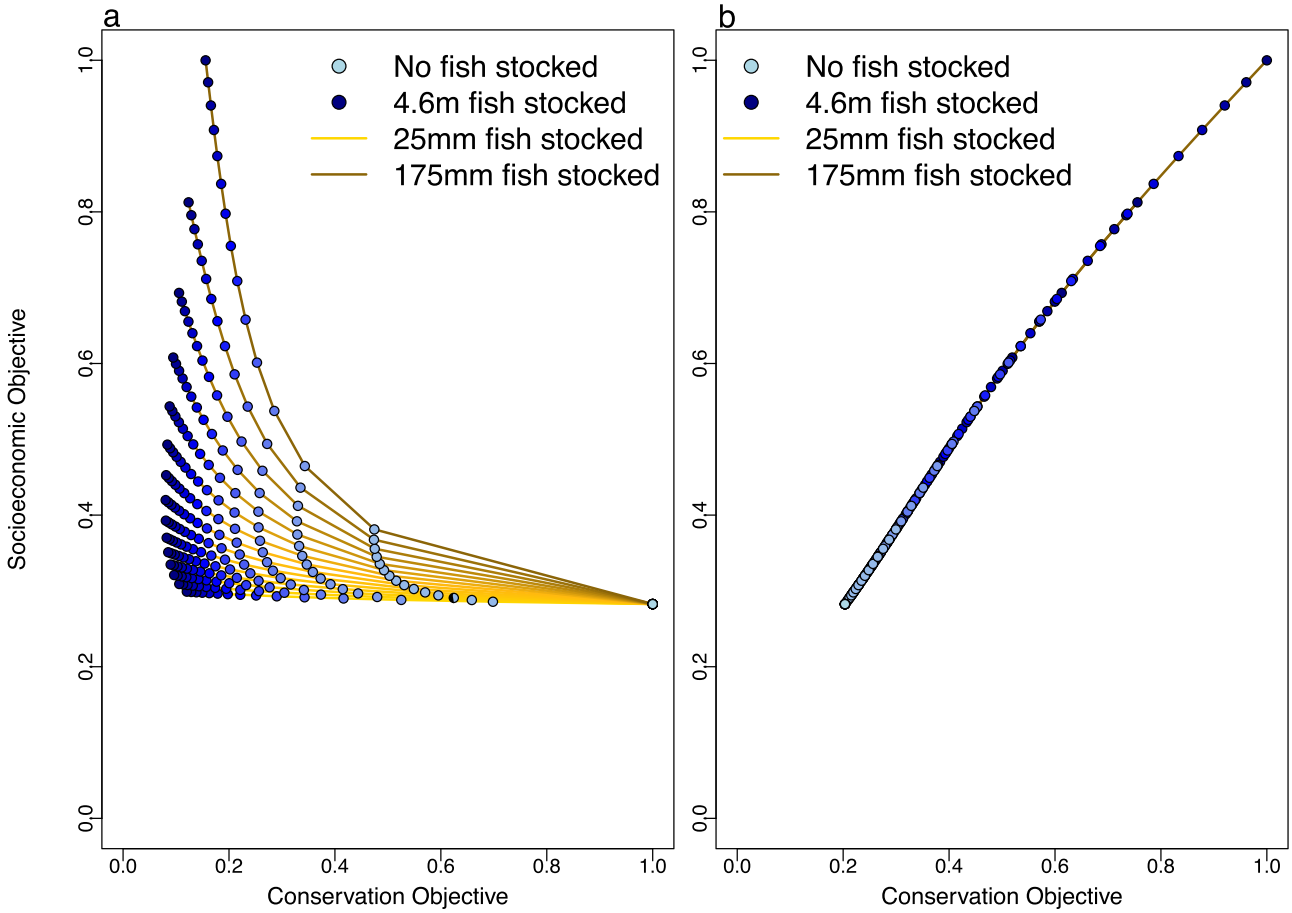

Fig. 1. Trade-off outcomes and associated frontier shapes resulting from alternative assumptions. Panel a shows points indicate relative values of conservation and socioeconomic objectives (0 is least, 1 is greatest) of alternative stocking scenarios with respect to size and number of fish stocked. Lighter colored points represent scenarios for which lesser numbers of fish were stocked and darker points represent scenarios with greater numbers of fish stocked (lightest blue = no fish stocked, darkest blue = $10 * \bar{R}_0$ fish stocked). Lines connecting points represent trade-off frontiers for a given size of fish stocked, with lighter colored lines representing smaller fish stocked and darker colors representing larger fish stocked (lightest = 25 mm, darkest = 175 mm total length). Panel b shows socioeconomic-conservation trade-off when considering that hatchery-type fish contribute to conservation objectives as wild-type fish do.

increases in socioeconomic objectives. Stocking increasing quantities of larger fish resulted in a deeply concave-up shape, such that great socioeconomic gains are possible, but only after substantial loss of conservation objectives (Fig. 1a, darkest lines). Stocking increasing quantities of small fish produced a nearly flat shape, indicating that stocking more fish provides essentially no increase in socioeconomic value but causes a substantial decline in conservation objectives (Fig. 1a, lightest lines). The alternative assumptions that hatchery fish were considered the equivalent of wild fish with respect to contribution to conservation objectives had obvious and dramatic effects on the trade-offs shapes (Fig. 1b). Under this assumption, the conservation-socioeconomic trade-off was eliminated, such that augmented socioeconomic objectives could be achieved concomitant to increases in conservation objectives. This occurred here due to the removal of a primary negative effect on the originally-formulated conservation objectives—the competition between wild fish and a hatchery population that grows with increased stocking. While there is still competition between stocked fish and the wild and hatchery populations, the number stocked each year is may be smaller (lighter colored points, Fig. 1b) relative to wild and hatchery components, such that competition is less important, or the size of the stocked fish may be large, which increases the probability of survival with minimal density dependent effects, benefiting both wild fish indirectly (via less competition) and hatchery fish directly (by augmenting their "spawning stock"). The effects of size at stocking are still evident, as smaller sized fish constitute lesser gains in socioeconomic and conservation objectives, relative to large fish (represented by the "thinning" of points moving upward on Fig. 1b).

3.2. Alternative objective functions

The intensity of trade-offs frontiers, but not the shapes themselves, changed under various assumptions of how catch-related satisfaction was related to catch rates (Fig. 2). When catch-related satisfaction was asymptotic (Fig. 2a), no stocking scenarios substantially increased socioeconomic objectives, though losses to conservation objectives were still realized and resulted in largely flat trade-off frontiers (Fig. 2b). This reflected the diminishing marginal returns to angler satisfaction from stocking-induced increases in catch rates, such that catch-related satisfaction, a key component of overall socioeconomic value, did not increase much even with large numbers of large fish stocked, compared to the initial assumptions of linearly increasing catch-related satisfaction with catch rates (Fig. 2c–d). Conversely, a relationship characterized by exponentially increasing catch-related satisfaction with increasing catch rates (Fig. 2e) produced greater socioeconomic objectives with the same conservation outcomes (Fig. 2f). This result occurred because angler catch-related satisfaction was assumed to exhibit marginal increases with greater catch rates under intense stocking, representing the idea that anglers are disproportionately satisfied with higher catch rates.

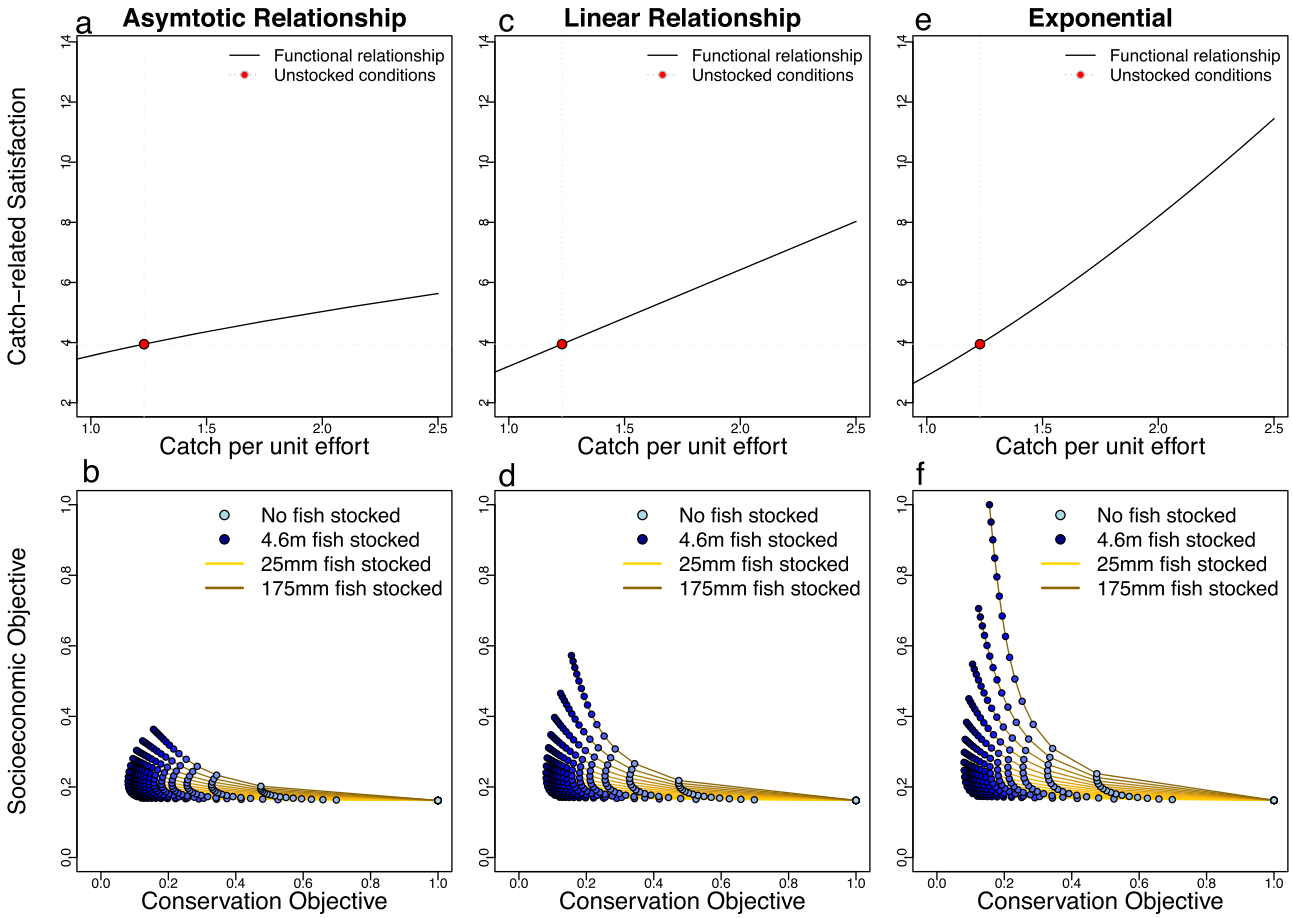

Fig. 2. Effects of alternative assumptions of catch-related satisfaction. Panels a, c, and e describe alternative assumptions of relationships between catch-related satisfaction experienced by anglers and nominal catch rates. Panels b, d, and f show the respective effects of these assumptions on socioeconomic-conservation trade-off outcomes and frontiers. Points and lines in panels b, d, and f are defined identically to those in Fig. 1.

Alternative assumptions of the relationship between equilibrium wild spawning biomass(B^*) and conservation objective value (CO) also provided differences in the steepness of trade-off frontier shapes (Fig. 3). If an asymptotic relationship between B^* and CO was assumed (per Equation (1)), a broad range of decreases in B^* resulted in a less-than-proportional declines in CO (Fig. 3a). This assumption revealed a right-shifted trade-off frontier, meaning that the greatest possible socioeconomic outcomes were achieved at substantially greater CO values, relative to the (initial) case where a linear relationship between CO and B^* (Fig. 3c–d). Additionally, the shapes of trade-off frontiers under this assumption varied depending on sizes of fish stocked. Smaller sizes still produced minimal socioeconomic gains and noticeable CO declines, but larger fish actually produced more linear trade-off frontier shapes with slopes suggesting that improvements in socioeconomic objectives were disproportionately greater than the CO losses. This occurred because greater catch rates and associated socioeconomic gains would be possible with larger fish stocked, as described in initial assumptions, but also because there was little loss to CO with initial declines in B^*. Further, the minimum CO achieved at the greatest number of fish stocked actually improves with larger sizes of fish stocked. This explanation is less intuitive. Larger fish would be nearer to the cessation of density dependent mortality when stocked and so would compete less with wild fish initially. Offspring of these larger stocked fish (hatchery fish) would compete with wild fish, but some proportion (Table 1, γ_h) also were assumed, through selective processes, to inherit traits allowing them to function as wild fish. While this process occurred in all previously described assumptions, the effects were generally minimal and scarcely noticeable on the trade-off frontiers (Fig. 1), but here (Fig. 3a–b) the relatively small gains in wild fish (B^*) associated with stocking larger fish coincide with the greatest changes of CO (Fig. 3a), such that small marginal increases in B^* associated with larger fish stocked are magnified to produce greater differences in CO. An alternative assumption of the B^* and CO relationship (per equation (2)) is that CO would begin to fall sharply as abundances of wild fish decrease below a threshold (Fig. 3f). This assumption yielded more extreme trade-off frontier shapes, with any gain in socioeconomic outcomes associated with a more severe CO loss (Fig. 3f). This occurred because—like many exploited fisheries—this red drum fishery is considered to be near the management threshold (B_R), such that even small declines in B^* resulting from stocking drive B^* below B_R and cause pronounced declines in CO.

3.3. Differential harvesting

Assumptions of alternative harvest regulations regarding origin of fish did little to alter the shapes of the trade-off frontiers (Fig. 4). Assuming that only wild and hatchery fish were harvest (Fig. 4a) produced more sever trade-off frontiers relative to the baseline assumptions, whereas assuming that only stocked fish would be harvested produced more moderate trade-offs. The relatively smaller changes produced by these alternative assumptions are related to the fact that changing the harvest regulations does nothing to alter the competition between wild, hatchery and stocked fish, which is a primary driver of trade-offs. Further, these

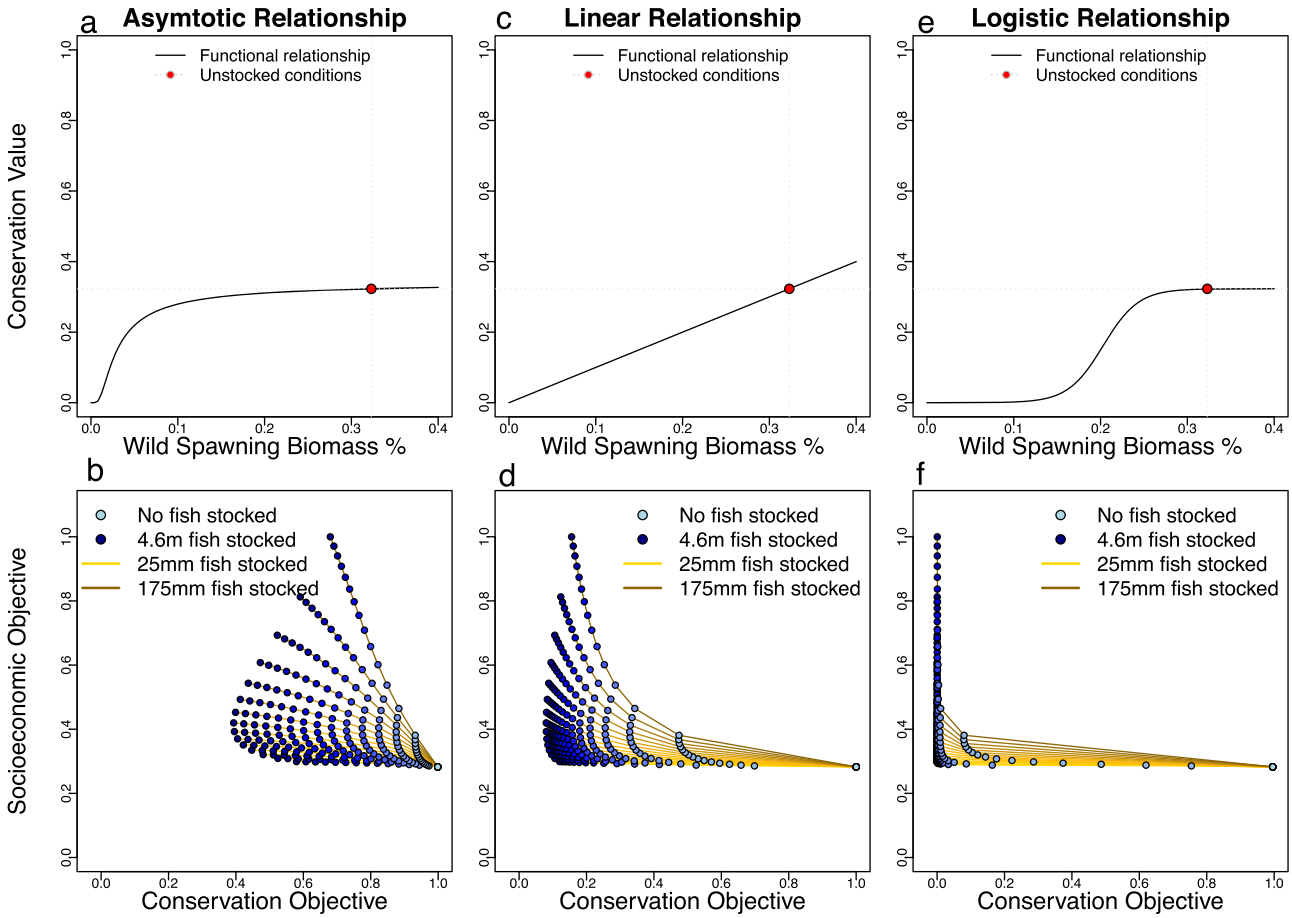

Fig. 3. Effects of alternative assumptions of the marginal conservation value of wild spawning biomass. First row panels (**a**, **c**, & **e**) describe alternative assumptions of functional relationships describing how diminishing proportions of wild spawning biomass affect relative conservation value. Second row panels (**b**, **d**, & **f**) show the respective effects of these assumptions on socioeconomic-conservation trade-off outcomes and frontiers. Points and lines in panels b, d, and f are defined identically to those in Fig. 1.

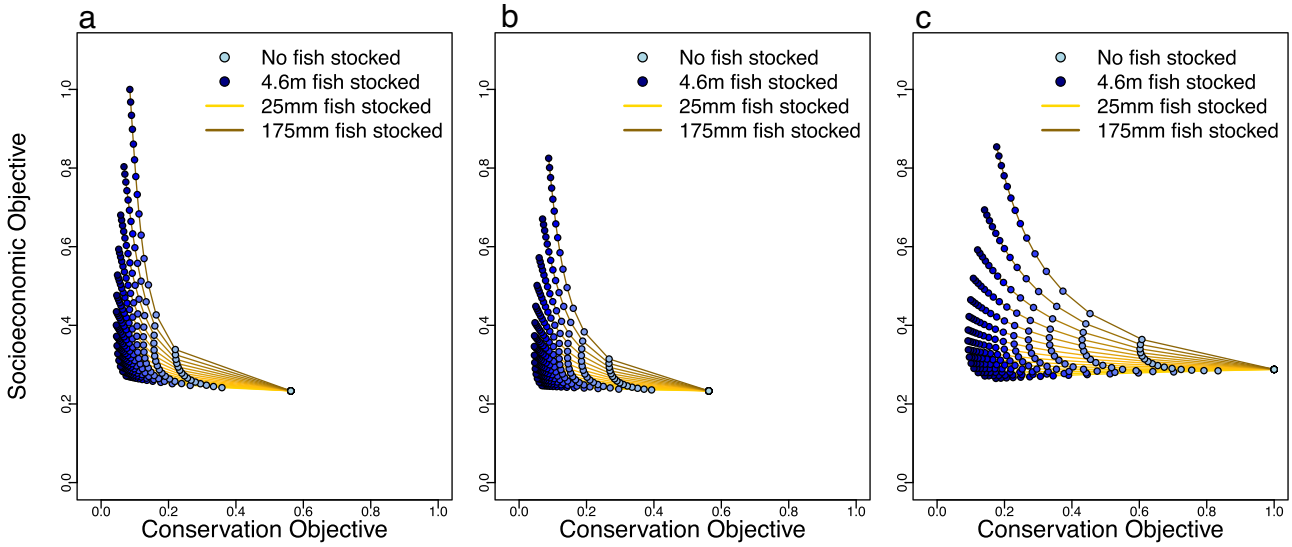

Fig. 4. Effect of population-component-specific harvest regulations on stock enhancement trade-offs. Panel a represents the assumption that only wild and hatchery fish are harvested. Panel b illustrates the initial assumptions that all fish (wild, hatchery and stocked) are harvested indiscriminately by anglers. Panel c represents the assumption that only stocked fish are harvested. For all scenarios, it is assumed that still many fish that are of legal harvest size are still released by anglers, as is likely in this fishery.

assumptions do not alter the composition of the trade-off frontiers themselves, as other alternative assumptions do (e.g., designation of hatchery fish as contributing to conservation objectives).

3.4. Inherent satisfaction from stocking

Assuming substantial inherent satisfaction from stocking (Fig. 5a) altered the actual shape of the trade-off frontier realized

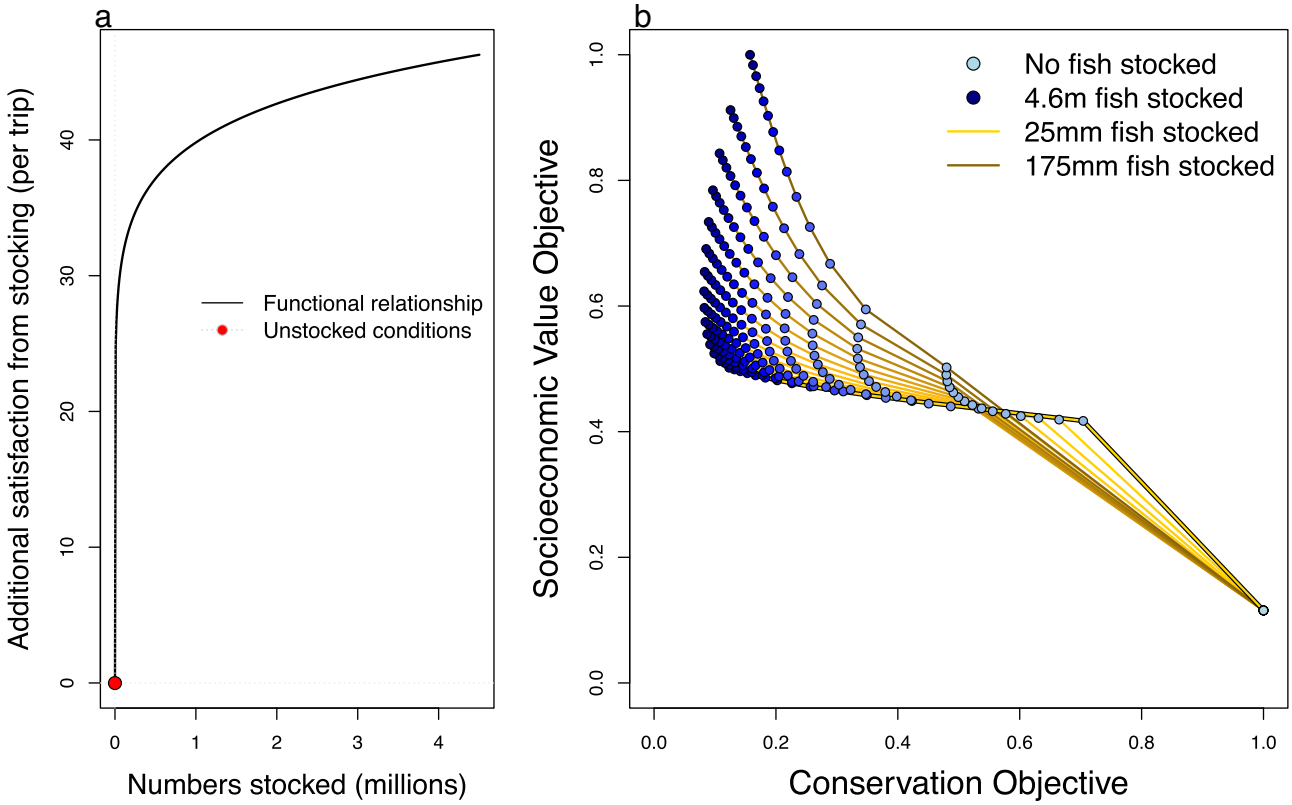

Fig. 5. Effects of assuming satisfaction is inherently related to stocking. Panel a describes an extreme example of how angler satisfaction could theoretically be inherently related to the act of stocking numbers of fish, and could additively increase the overall socioeconomic objectives. Panel b describes the effect of this assumption on socioeconomic-conservation trade-off outcomes and frontiers. Points and lines in panel b are defined identically to those in Fig. 1, with the addition of black outlines on the frontier associated with the smallest size of fish stocked (25 mm) for clarity.

under stocking (Fig. 5b). The resultant concave trade-off frontier suggests an optimal (relative to alternatives) scenario would be stocking minimal numbers of small fish. Stocking in this way would do little to increase the overall population of fish (since small fish were would be subject to nearly all density dependent mortality processes) and would cause minimal replacement of wild fish (since few were stocked). Essentially, this would be "mock stocking" or purposefully stocking in such a way to attain greater inherent stakeholder satisfaction from operating a stocking program, but not to cause any real biological effects.

4. Discussion

This study suggests that stock enhancement is not as a technological panacea (Holling and Meffe, 1996; van Poorten et al., 2011) that can circumvent or solve trade-offs between fundamental conservation and socioeconomic objectives of recreational fisheries management, particularly where the abundance of truly wild fish is central to the conservation objectives. In that case, mutually successful "compromises" between these objectives will be generally difficult to achieve with stock enhancement, owing to the nature of the trade-off frontiers. The steeply concave-up shape of the trade-off frontiers under most assumptions suggests the opportunity cost (in terms of conservation objectives) is very high relative to gains in socioeconomic objectives, at least until the conservation objective is largely dissipated. In terms of management actions, this means that the stocking scenarios most likely to yield meaningful increases in socioeconomic objectives (i.e. stocking greater numbers of larger fish) are precisely those that would be expected to cause substantial declines in conservation objectives, and conversely, stocking scenarios most useful for conserving wild populations (i.e. not stocking at all, or stocking lesser numbers of small fish) forgo substantial socioeconomic gains. This information has immediate implications for the management of wild and enhanced recreational fisheries: stock enhancement does not well serve socioeconomic and conservation objectives simultaneously in recreational fisheries wherever maintenance of wild populations is considered important. Trade-offs are alleviated, however, when naturally recruited hatchery-type fish are considered equivalent to wild fish in conservation value. This result highlights the importance of clearly specifying conservation objectives, including the value of wild and naturally recruited hatchery fish when assessing stock enhancements. Different formulations of conservation objectives imply very different tradeoffs, a fact increasingly appreciated in the management of Pacific salmon fisheries (Naish et al., 2007; Paquet et al., 2011 Paquet et al., 2011).

The findings of this study suggesting stock enhancement poses risks to wild populations are corroborated by previous studies (Hilborn and Eggers, 2000; Bohlin et al., 2002; Camp et al., 2014), but the novelty of this work lies in the assertion that such risks are nearly inherent if socioeconomic objectives are to be augmented through the positive effect of stocking on fish populations and conservation objectives are focused entirely on truly wild fish. Improved socioeconomic outcomes of recreational fisheries from enhancement usually require increased angler effort, increased catch rates, or both by increasing the population of catchable fish (Camp et al., 2013). Such increases will almost tautologically negatively affect wild fish populations in the form of attracted angler effort (Baer and Brinker, 2010) or competition with stocked and naturally recruited hatchery-type fish (Naish et al., 2007). Thus the steep, concave-up nature of this inverse relationship in most trade-off frontiers described in this work (e.g., Fig. 1) occurs for

two primary reasons: (1) increases in socioeconomic outcomes requires stocking larger fish, and little socioeconomic value related to angler catch rates is generated by stocking very few or very small fish (Camp et al., 2014), and (2) any substantial increase in overall abundance of a fish population that is not recruitment overfished will entail partial replacement of wild with stocked fish due to compensatory density-dependence in the pre-recruit stage (Lorenzen, 2005). Differential harvesting of stocked or wild fish produced muted effects relative to alternative assumptions about objectives and their composition. This suggests that while differential regulations could mitigate conservation losses, they should not be expected to dramatically alter the trade-offs in recreational fisheries with overall sustainable harvest levels and a high incidence of voluntary releases.

While the socioeconomic-conservation trade-offs predicted here follow from recruitment and recreational fisheries theory, there are several reasons they may not yet be well recognized by management. Stocking fish large enough to bypass much of the density-dependent-survival during the recruitment period is critical to realizing higher overall fish abundance and socioeconomic objectives (Leber et al., 2005), but many stock enhancement programs stock small fish that are likely susceptible to extensive density dependent mortality (McEacheron et al., 1998; Tringali et al., 2008). Such stocking would be expected to result in some displacement of wild fish (conservation losses) but no real increase in socioeconomic value (Hilborn and Eggers, 2000). If stocked fish are unmarked and not distinguishable from wild fish, such replacement may be cryptic (Lorenzen et al., 2010). If larger fish are stocked, they may not be stocked in sufficient numbers to lead to a noticeable increase in abundance or catch rates owing to generally high variation in natural recruitment of wild fish. Finally, despite the long history of stock enhancement in recreational fisheries, quantitative evaluations of the population effects of this strategy are rare (Hilborn, 1992; Taylor et al., 2005; Camp et al., 2013). Better monitoring of enhancement will be crucial in future work designed to empirically test the predicted shape and nature of trade-offs.

Even though stocking is generally unlikely to avoid fundamental trade-offs between conservation and socioeconomic objectives in recreational fisheries, analyses of the trade-off frontiers revealed some specific scenarios exist in which enhancement could achieve socioeconomic gains at lower conservation costs. Perhaps the most obvious occurs when hatchery-type fish are considered of equivalent conservation value as wild fish (Fig. 1b). A similar situation occurs when abundance of wild fish is of little importance to conservation objectives, as described by the assumption that the conservation objective declines substantially only at very low wild spawning biomass levels (Fig. 3a–b). While such-formulated conservation objectives may eventually risk the resilience of the socioecological system (Holling and Meffe, 1996), our work demonstrates socioeconomic gains can be had in such systems, at least temporarily. Greater socioeconomic value at lesser impacts to conservation objectives could also be achieved when inherent satisfaction from stocking was assumed and very small numbers of small fish were stocked (Fig. 5). This type of enhancement would have a low probability of increasing overall fish abundance or catch rates, both of which are traditionally prerequisites for achieving socioeconomic objectives of stock enhancement (Camp et al., 2013), but rather would increase socioeconomic objectives by appealing to stakeholders' hypothetical affinity for enhancement programs. While a recent study has suggested angler intrinsic preference for stocking in minimal, this has not been widely investigated, particularly in open access marine fisheries (Arlinghaus et al., 2014). Rather than suggesting that such a strong inherent satisfaction response is likely, the broader implication of this result is to emphasize consideration of the effect management strategies themselves directly have on stakeholder satisfaction, in addition to what they may contribute by affecting the resource (Chu et al., 2010). Management strategies designed to address stakeholder well-being directly, whether it be implementing desired strategies or improving fishing access or facilities, may be more efficient than those working indirectly through altering stock levels (e.g., stock enhancement), particularly if the latter involve biological uncertainty (e.g., survival of stocked fish) or introduce risk to conservation objectives. This is considered a verdant area for future research. A final scenario described by this work in which stocking could achieve socioeconomic objectives at a low opportunity costs occurs when the conservation objective has been substantially compromised. In our case study this is represented near the point (0.4, 0.4) of Fig. 1a. Here, additional intense stocking of larger fish could augment socioeconomic objectives at decreased marginal costs to conservation objectives. Managing for such low conservation outcomes might not be viable region-wide, but in smaller spatial areas, "sacrificing" conservation objectives to attain greater socioeconomic value may be acceptable, subject to concerns over spatial equity (Halpern et al., 2011). Such spatially diversified strategies with respect to management objectives and actions also represent a topic ripe for future research (Lester et al., 2003; Lester et al., 2013).

How stakeholders and society in general consider wild, hatchery and stocked fish, as well as overall fisheries management objectives, has substantial influence on our work. The strongest conservation-socioeconomic tradeoffs identified here occur under the assumption that only the wild population component contributes to the conservation objective. This assumption is supported by observed differences between wild and hatchery fish (Araki et al., 2008; Fraser, 2008) and in certain fisheries, angler preferences (Olaussen and Liu, 2011). In some situations however, the stocked and hatchery (naturally recruited fish of hatchery parentage) components may be viewed as contributing to conservation objectives but at a lower weight than the wild stock component. This may be the case, for example, where hatchery production results in fish that are indistinguishable from wild fish in their genetic makeup and fitness (a feat that can be achieved in some conservation hatchery programs, but is typically very expensive; Lorenzen et al., 2012), or where the wild population is deemed of limited conservation interest (e.g., an introduced species). Under such scenarios the tradeoffs were substantially less severe (Fig. 1b). With respect to the socioeconomic objectives, this work considers the value of a captured fish to an angler to be identical for stocked, hatchery and wild fish. While this has rarely been assessed, Anderson and Lee (2013) and Olaussen and Liu (2011) found this value could differ in Pacific and Norwegian salmon fisheries, respectively. In reality, most recreational fisheries will be characterized by stakeholders with heterogeneous preferences for system services (Breffle and Morey, 2000; Schuhmann and Schwabe, 2004), whether relating to hatchery versus wild fish or the demand for additional catch. While this work explores some alternative assumptions of how stakeholders as a population would respond (e.g., catch-related satisfaction, inherent satisfaction from stocking), we do not explore how variability within the group of stakeholders would influence results, and this also represents an area for future work.

A critical element of bioeconomic models of recreational fisheries is the representation of angler behavior and satisfaction (e.g., Fenichel et al., 2013), and the simple representations in this work have implications for the inferences drawn. In all analyses here, aggregate fishing effort (which links the biological and socioeconomic sub-models) is assumed related to vulnerable fish abundance. Of course, other factors (e.g., travel cost, site attributes, species available, etc.) can and do influence individual angler choices regarding how much to fish (Schuhmann and Schwabe, 2004; Hunt, 2005; Haab et al., 2012). However, the approach taken in this work has precedent where research focuses on understanding the effect of management strategies at the population-level—as

opposed to individual choice (Cox and Walters, 2002; Allen et al., 2013). Additionally, Camp et al. (2016) showed the best available data describing aggregate red drum recreational fishing effort in Florida could be well predicted by estimated abundance of vulnerable fish. A separate and more implicit assumption of this work is that changes in red drum catch rate are directly related to changes in overall angler satisfaction. This assumption is reasonable given positive associations between red drum catch rates and angler benefits that have been empirically shown (Schuhmann, 1998; Haab et al., 2012), provided one is also willing to assume other factors contributing to catch-related utility remain largely unchanged by stocking. One such factor usually important to angler satisfaction is the somatic size of fish caught (Arlinghaus et al., 2014). However, Florida red drum are primarily vulnerable to recreational capture for a brief temporal (approximately on year) and narrow size (approximately 0.4–0.75m) window as sub-adults owing to ontogenetic movement patterns (Murphy and Muyandorero, 2009), such that even if stocking (and therefore density) affected the length-at-infinity size of red drum, as has been shown for some species (Lorenzen and Enberg, 2002; Dotson et al., 2013), this would not likely alter the sizes of red drum captured by anglers. However, if other factors change as a function of stocking (e.g., crowding of anglers) or if catch-related satisfaction is more or less important (i.e., lower weight δ in Table 2, E.40) the socioeconomic value calculated here could be over- or under-estimated. These contingencies should preclude drawing any inferences from this work regarding absolute socioeconomic value that could be generated from stocking in this system.

There are other elements of the socioecological recreational fisheries system that could affect trade-offs realized with enhancement that we do not account for with this model. First, we present equilibrium results expected under average conditions using a phenomenological approach, as opposed to a mechanistic representation of angler decision-making (Hunt et al., 2007; Abbott and Fenichel, 2013), or presenting time-dynamic streams of value. Similarly, further dynamics of the governance process—either feedback loops whereby hatchery operations might be modified as monitoring or fiscal information (e.g., production efficiency of the facility or costs for plant upgrades) is received, or the potential benefits to governance of one strategy over another. Our work also does not explicitly consider financial costs of stock enhancement, which would, *ceteris paribus*, likely increase with size and numbers of fish stocked and so would erode some of the socioeconomic gains shown possible with these implementations. Finally, our study refers to trade-offs in a single spatial area that is hypothetically managed without regard to other areas. This may be common in certain situations (e.g., private angling clubs, community-based resource reef fisheries, etc.) where conservation and socioeconomic objectives must be obtained from essentially the same spatial region, but more open-access systems with more complex spatial planning and management may be better represented in the future with spatially explicit modeling efforts. While the assumptions of this study may limit inferences regarding absolute socioeconomic or conservation value, they also allow for a relatively simple and generalizable assessment of socioeconomic-conservation trade-offs realized with stock enhancement, and in doing so make more explicit the expected outcomes of this common management action.

5. Conclusions

This study employed a stylized model of a fishery with wild, stocked and hatchery fish to evaluate conservation and socioeconomic tradeoffs under different assumptions of angler behavior and preferences. The analysis yielded several general and case-specific conclusions. Generally, evaluating trade-offs with quantitative models may be useful for abetting management discussions by depicting alternatives with mutually exclusive outcomes. These models are likely to be most useful when various stakeholder opinions and management preferences are explicitly considered. Specific to our case study of recreational stock enhancement, three primary messages emerge: (1) nearly any recreational stock enhancement is likely to have high opportunity costs in terms of reducing populations of truly wild fish, (2) the trade-off frontier associated with stocking is robust to many realistic assumptions and makes optimal "compromises" between wild-fish conservation and socioeconomic objectives unlikely, but (3) stock enhancement could still be a useful strategy where naturally-recruited hatchery-type fish are regarded as similar to wild fish in conservation value, where the abundance of wild fish is not considered important, or if stakeholders (or managers) desire stocking programs for other, inherent reasons. Future studies could utilize information from stated preference studies related to stocking programs from recreational anglers, non-anglers and managers in order to integrate additional benefits associated with stocking programs and more accurately reflect costs over time, especially if new facilities or technologies are planned.

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References

Abbott, J.K., Fenichel, E.P., 2013. Anticipating adaptation: a mechanistic approach for linking policy and stock status to recreational angler behavior. Can. J. Fish Aquat. Sci. 70, 1190–1208.

Allen, M.S., Ahrens, R.N.M., Hansen, M.J., Arlinghaus, R., 2013. Dynamic angling effort influences the value of minimum length limits to prevent recruitment overfishing. Fish. Manage. Ecol. 20, 247–257.

Anderson, L.G., 1993. Toward a complete economic theory of the utilization and management of recreational fisheries. J. Environ. Econ. Manage. 24, 272–295.

Anderson, L.E., Lee, S.T., 2013. Untangling the recreational value of wild and hatchery salmon. Mar. Resour. Econ. 28 (2), 175–197.

Araki, H., Berejikian, B.A., Ford, M.J., Blouin, M.S., 2008. Fitness of hatchery-reared salmonids in the wild. Evol. Appl. 1, 342–355.

Arlinghaus, R., 2005. A conceptual framework to identify and understand conflicts in recreational fisheries systems, with implications for sustainable management. Aquat. Resour. Cult. Dev. 1, 145–174.

Arlinghaus, R., Beardmore, B., Riepe, C., Meyerhoff, J., Pagel, T., 2014. Species-specific preferences of German recreational anglers for freshwater fishing experiences, with emphasis on intrinsic utilities of fish stocking and wild fishes. J. Fish Biol. 85, 1843–1867.

Baer, J., Brinker, A., 2010. The response of a brown trout stocks and perception of anglers to cessation of brown trout stocking. Fish. Manage. Ecol. 17, 157–164.

Beardmore, B., Hunt, L.M., Haider, W., Dorow, M., Arlinghaus, R., 2015. Effectively managing angler satisfaction in recreational fisheries requires understanding the fish species and the anglers. Can. J. Fish Aquat. Sci. 72, 500–513.

Bell, J.D., Leber, K.M., Blankenship, H.L., Loneragan, N.R., Masuda, R., 2008. A new era for restocking, stock enhancement and sea ranching of coastal fisheries resources. Rev. Fish. Sci. 16, 1–9.

Bohlin, T., Sundström, L.F., Johnsson, J.I., Höjesjö, J., Pettersson, J., 2002. Density-dependent growth in brown trout: effects of introducing wild and hatchery fish. J. Anim. Ecol. 71, 683–692.

Breffle, W.S., Morey, E.R., 2000. Investigating preference heterogeneity in a repeated discrete-choice recreation demand model of Atlantic salmon fishing. Mar. Resour. Econ. 15, 1–20.

Camp, E.V., Lorenzen, K., Ahrens, R., Barbieri, L., Leber, K.M., 2013. Potentials and limitations of stock enhancement in marine recreational fisheries systems: an integrative review of Florida's red drum enhancement. Rev. Fish. Sci. 21, 388–402.

Camp, E.V., Lorenzen, K., Ahrens, R., Allen, M.S., 2014. Stock enhancement to address multiple recreational fisheries objectives: an integrated model applied to red drum Sciaenops ocellatus in Florida. J. Fish Biol. 85 (6), 1868–1889.

Camp, E.V., Poorten, B.T.V., Walters, C.J., 2015. Evaluating short openings as a management tool to maximize catch-related utility in catch-and-release fisheries. N. Am. J. Fish. Manage. 35 (6), 1106–1120.
Camp, E.V., Ahrens, R., Allen, M.S., Lorenzen, K., 2016. Relationships between angling effort and fish abundance in recreational marine fisheries. Fish. Manage. Ecol., http://dx.doi.org/10.1111/fme.12168.
Cheung, W.W., Sumaila, U.R., 2008. Trade-offs between conservation and socio-economic objectives in managing a tropical marine ecosystem. Ecol. Econ. 66, 193–210.
Chu, J., Anderson, J.L., Asche, F., Tudur, L., 2010. Stakeholders' perceptions of aquaculture and implications for its future: a comparison of the USA and Norway. Mar. Resour. Econ. 25 (1), 61–76.
Coggins, L.G., Catalano, M.J., Allen, M.S., Pine, W.E., Walters, C.J., 2007. Effects of cryptic mortality and the hidden costs of using length limits in fishery management. Fish Fish. 8 (3), 196–210.
Cooke, S.J., Arlinghaus, R., Johnson, B.M., Cowx, I.G., 2015. Recreational fisheries in inland waters. In: Craig, J.F. (Ed.), Freshwater Fisheries Ecology. John Wiley & Sons, Ltd., Chichester, UK, pp. 449–465.
Cowx, I.G., Arlinghaus, R., Cooke, S.J., 2010. Harmonizing recreational fisheries and conservation objectives for aquatic biodiversity in inland waters. J. Fish. Biol. 76, 2194–2215.
Cox, S.P., Walters, C., 2002. Modeling exploitation in recreational fisheries and implications for effort management on British Columbia rainbow trout lakes. N. Am. J. Fish. Manage. 22, 21–34.
Cox, S.P., Walters, C.J., Post, J.R., 2003. A model-based evaluations of active management of recreational fishing effort. N. Am. J. Fish. Manage. 23, 1294–1302.
Denson, M.R., Brenkert, I.V.K., Jenkins, W.E., Darden, T.L., 2012. Assessing red drum juvenile stocking in a South Carolina estuary using genetic identification. N. Am. J. Fish. Manage. 32 (1), 32–43.
Dorow, M., Beardmore, B., Haider, W., Arlinghaus, R., 2010. Winners and losers of conservation policies for European Eel: *Anguilla anguilla*: an economic welfare analysis for differently specialised eel anglers. Fish. Manage. Ecol. 17, 106–125.
Dotson, J.R., Allen, M.S., Kerns, J.A., Pouder, W.F., 2013. Utility of restrictive harvest regulations for trophy largemouth bass management. N. Am. J. Fish. Manage. 33 (3), 499–507.
Fenichel, E.P., Abbott, J.K., Huang, B., 2013. Modelling angler behaviour as a part of the management system: synthesizing a multi-disciplinary literature. Fish Fish. 14 (2), 137–157.
Figge, F., 2004. Bio-folio: applying portfolio theory to biodiversity. Biodivers. Conserv. 13, 827–849.
Fraser, D.J., 2008. How well can captive breeding programs conserve biodiversity? A review of salmonids. Evol. Appl. 1, 535–586.
Gaydon, D.S., Meinke, H., Rodriguez, D., McGrath, D.J., 2012. Comparing water options for irrigation farmers using modern portfolio theory. Agric. Water Manage. 115, 1–9.
Haab, T., Hicks, R., Schnier, K., Whitehead, J.C., 2012. Angler heterogeneity and the species-specific demand for marine recreational fishing. Mar. Resour. Econ. 27 (3), 229–251.
Halpern, B.S., White, C., Lester, S.E., Costello, C., Gaines, S.D., 2011. Using portfolio theory to assess tradeoffs between return from natural capital and social equity across space. Biol. Conserv. 144 (5), 1499–1507.
Halverson, M.A., 2008. Stocking trends: a quantitative review of governmental fish stocking in the United States, 1931 to 2004. Fish 33 (2), 69–75.
Ham, K.D., Pearsons, T.N., 2001. A practical approach for containing ecological risks associated with fish stocking programs. Fish 26 (4), 15–23.
Hilborn, R., 1992. Hatcheries and the future of salmon in the Northwest. Fish 17 (1), 5–8.
Hilborn, R., 1998. The economic performance of marine stock enhancement projects. Bull. Mar. Sci. 62 (2), 661–674.
Hilborn, R., 2007. Defining success in fisheries and conflicts in objectives. Mar. Policy 31, 153–158.
Hilborn, R., Eggers, D., 2000. A review of the hatchery programs for pink salmon in Prince William Sound and Kodiak Island, Alaska. Trans. Am. Fish. Soc. 129 (2), 333–350.
Holling, C.S., Meffe, G.K., 1996. Command and control and the pathology of natural resource management. Conserv. Biol. 10 (2), 328–337.
Hunt, L.M., 2005. Recreational fishing site choice models: insights and future opportunities. Hum. Dimens. Wildl. 10, 153–172.
Hunt, L.M., Boxall, P.C., Boots, B., 2007. Accommodating complex substitution patterns in a random utility model of recreational fishing. Mar. Resour. Econ. 22, 155–172.
Koehn, J., 2010. Conservation and utilization: harnessing forces for better outcomes. Ecol. Manage. Restor. 11, 86–87.
Koehn, J.D., Todd, C.R., 2012. Balancing conservation and recreational fishery objectives for a threatened fish species, the Murray cod, *Maccullochella peelii*. Fish. Manage. Ecol. 19 (5), 410–425.
Lackey, R.T., 1998. Fisheries management: integrating societal preference, decision analysis, and ecological risk assessment. Environ. Sci. Policy 1 (4), 329–335.
Lackey, R.T., 2003. Setting goals and objectives in managing for healthy ecosystems. Manag. Healthy Ecosyst., 165–166.
Lackey, R.T., 2004. Societal values and the proper role of restoration ecologists. Front. Ecol. Environ. 2 (2), 45–46.
Leber, K.M., Cantrell, R.N., Leung, P.S., 2005. Optimizing cost effectiveness of size at release in stock enhancement programs. N. Am. J. Fish. Manage. 25, 1596–1608.
Lester, N.P., Marshall, T.R., Armstrong, K., Dunlop, W.I., Ritchie, B., 2003. A broad-scale approach to management of Ontario's recreational fisheries. N. Am. J. Fish. Manage. 23 (4), 1312–1328.
Lester, S.E., Costello, C., Halpern, B.S., Gaines, S.D., White, C., Barth, J.A., 2013. Evaluating tradeoffs among ecosystem services to inform marine spatial planning. Mar. Policy 38, 80–89.
Lorenzen, K., 2005. Population dynamics and potential of fisheries stock enhancement: practical theory for assessment and policy analysis. Philos. Trans. R. Soc. Br. B 360, 171–189.
Lorenzen, K., 2008. Understanding and managing enhancement fisheries systems? Rev. Fish. Sci. 16 (1-3), 10–23.
Lorenzen, K., 2014. Understanding and managing enhancements: why fisheries scientists should care. J. Fish Biol. 85 (6), 1807–1829.
Lorenzen, K., Enberg, K., 2002. Density-dependent growth as a key mechanism in the regulation of fish populations: evidence from among-population comparisons. Proc. R. Soc. Lon. B: Biol. Sci. 269 (1486), 49–54.
Lorenzen, K., Leber, K.M., Blankenship, H.L., 2010. Responsible approach to marine stock enhancement: an update. Rev. Fish. Sci. 18, 198–210.
Lorenzen, K., Beveridge, M., Mangel, M., 2012. Cultured fish: integrative biology and management of domestication and interactions with wild fish. Biol. Rev. 87 (3), 639–660.
Mardle, S., Pascoe, S., 2002. Modelling the effects of trade-offs between long and short-term objectives in fisheries management. J. Environ. Manage. 65 (1), 49–62.
McClenachan, L., 2013. Recreation and the right to fish movement: anglers and ecological degradation in the Florida Keys. Environ. Hist. 18, 76–87.
McConnell, K.E., Sutinen, J.G., 1979. Bioeconomic models of marine recreational fishing. J. Environ. Econ. Manage. 6 (2), 127–139.
McEacheron, L.W., Colura, R.C., Bumguardner, B.W., Wood, R., 1998. Survival of stocked red drum in Texas. Bull. Mar. Sci. 62, 359–368.
McNie, E.C., 2007. Reconciling the supply of scientific information with user demands: an analysis of the problem and review of the literature. Environ. Sci. Policy 10, 17–38.
Murphy, M.D., Muyandorero, J., 2009. A Stock Assessment of Red Drum, Sciaenops Ocellatus in Florida: Status of the Stocks Through 2007. Report 2008-008 to the Florida Fish and Wildlife Conservation Commission. Fish and Wildlife Research Institute, St. Petersburg, FL.
Myers, R.A., Levin, S.A., Lande, R., James, F.C., Murdoch, W.W., Paine, R.T., 2004. Hatcheries and endangered salmon. Sciene 303 (5666), 1980.
Naish, K.A., Taylor, J.E., Levin, P.S., Quinn, T.P., Winton, J.R., Huppert, D., Hilborn, R., 2007. An evaluation of the effects of conservation and fishery enhancement hatcheries on wild populations of salmon. Adv. Mar. Biol. 53, 61–194.
National Marine Fisheries Service (NMFS), 2013. Personal Communication. Fisheries Statistics Division (accessed 20.04.13.).
Olaussen, J.O., Liu, Y., 2011. On the willingness-to-pay for recreational fishing—escaped farmed versus wild Atlantic Salmon. Aquacult. Econ. Manage. 15 (4), 245–261.
Paquet, P.J., Flagg, T., Appleby, A., Barr, J., Blankenship, L., Campton, D., et al., 2011. Hatcheries, conservation, and sustainable fisheries—achieving multiple goals: results of the Hatchery Scientific Review Group's Columbia River basin review. Fish 36 (11), 547–561.
Possingham, H., Shea, K., 1999. The Business of Biodiversity, 31. Aust. Zoo, pp. 3–5.
Propst, D.B., Gavrilis, D.G., 1987. Role of economic impact assessment procedures in recreational fisheries management. Trans Am. Fish. Soc. 116 (3), 450–460.
Quinn, T.P., Kinnison, M.T., Unwin, M.J., 2001. Evolution of chinook salmon (*Oncorhynchus tshawytscha*) populations in New Zealand: pattern, rate, and process. Genetica 112 (1), 493–513.
Richards, R.A., Rago, P.J., 1999. A case history of effective fishery management: Chesapeake Bay striped bass. N. Am. J. Fish Manage. 19 (2), 356–375.
Rogers, M.W., Allen, M.S., Brown, P., Hunt, T., Fulton, W., Ingram, B.A., 2010. A simulation model to explore the relative value of stock enhancement versus harvest regulations for fishery sustainability. Ecol. Model 221 (6), 919–926.
Sanchirico, J.N., Smith, M.D., Lipton, D.W., 2008. An empirical approach to ecosystem-based fishery management. Ecol. Econ. 64, 586–596.
Scharf, F.S., 2000. Patterns in abundance, growth and mortality of juvenile red drum across estuaries in Texas coast with implications for recruitment and stock enhancement. Trans. Am. Fish. Soc. 129, 1207–1222.
Schuhmann, P.W., 1998. Deriving species-specific benefits measures for expected catch improvements in a random utility framework. Mar. Resour. Econ. 13, 1–21.
Schuhmann, P.W., Schwabe, K.A., 2004. An analysis of congestion measures and heterogeneous angler preferences in a random utility model of recreational fishing. Environ. Res. Econ. 27 (4), 429–450.
Shea, K., 1998. Management of populations in conservation, harvesting and control. Trends Ecol. Evol. 3 (9), 371–375.
Sullivan, M.G., 2003. Active management of Walleye fisheries in Alberta: dilemmas of managing recovering fisheries. N. Am. J. Fish. Manage. 23, 1343–1358.
Sylvia, G., Cai, D., 1995. Generating policy information for fisheries management: a comparison of alternative approaches. Mar. Resour. Econ. 10 (1), 77–91.
Sylvia, G., Enriquez, R.R., 1994. Multiobjective bioeconomic analysis: an application to the Pacific whiting fishery. Mar. Resour. Econ. 9 (4), 311–328.
Taylor, M.D., Palmer, P.J., Fielder, D.S., Suthers, I.M., 2005. Responsible estuarine finfish stock enhancement: an Australian perspective. J. Fish Biol. 67 (2), 299–331.
Tringali, M.D., Leber, K.M., Halstead, W.G., McMichael, R., O'Hop, J., Winner, B., Cody, R., Young, C., Neidig, C., Wolfe, H., Forstchen, A., Barbieri, L., 2008. Marine

stock enhancement in Florida A multi-disciplinary, stakeholder-supported, accountability-based approach. Rev. Fish. Sci. 16, 51–57.
van Poorten, B.T., Arlinghaus, R., Daedlow, K., Haertel-Borer, S.S., 2011. Social-ecological interactions, management panaceas, and the future of wild fish populations. Proc. Natl. Acad. Sci. U. S. A. 108, 12544–12559.
Vega, R., 2011. Enhancement of Texas sciaenids (red drum and spotted seatrout). Texas, October 26–27, 2009. U.S. Dept. Commerce, NOAA Tech. Memo. NMFS-F/SPO-113 In: Stickney, R., Iwamoto Rust, R.M. (Eds.), Interactions of Fisheries and Fishing Communities Related to Aquaculture: Proceedings of the Thirty-eighth U.S.—Japan Aquaculture Panel Symposium, Corpus Christi, 2011, pp. 85–92.
Walters, C.J., Martell, S.J.D., 2004. Fisheries Ecology and Management. Princeton University Press, Princeton.
Weiss, S., Schmutz, S., 1999. Performance of hatchery-reared brown trout and their effects on wild fish in two small Austrian streams. Trans Am. Fish. Soc. 128 (2), 302–316.