Marine Policy 67 (2016) 105–117

Contents lists available at ScienceDirect

Marine Policy

journal homepage: www.elsevier.com/locate/marpol

Designing and financing optimal enforcement for small-scale fisheries and dive tourism industries

Gavin McDonald *[,1], Tracey Mangin [1], Lennon R. Thomas [1], Christopher Costello

Bren School of Environmental Science and Management, University of California, Santa Barbara, CA 93106, USA

ARTICLE INFO

Article history:
Received 8 August 2015
Received in revised form
1 February 2016
Accepted 2 February 2016
Available online 11 February 2016

Keywords:
Optimal fisheries enforcement
Fisheries cost-recovery
Ecosystem services
Illegal, unregulated, and unreported (IUU) fishing
Small-scale fisheries

ABSTRACT

Effective enforcement can reduce the impacts of illegal, unregulated, and unreported (IUU) fishing, resulting in numerous economic, ecological, and social benefits. However, resource managers in small-scale fisheries often lack the expertise and financial resources required to design and implement an effective enforcement system. Here, a bio-economic model is developed to investigate optimal levels of fishery enforcement and financing mechanisms available to recover costs of enforcement. The model is parameterized to represent a small-scale Caribbean lobster fishery, and optimal fishery enforcement levels for three different stakeholder archetypes are considered: (1) a fishing industry only; (2) a dive tourism industry only; and (3) fishing and dive tourism industries. For the illustrative small-scale fishery presented, the optimal level of fishery enforcement decreases with increasing levels of biomass, and is higher when a dive tourism industry is present. Results also indicate that costs of fisheries enforcement can be recovered through a suite of financing mechanisms. However, the timescale over which financing becomes sustainable will depend largely on the current status of the fishery resource. This study may serve as a framework that can be used by resource managers to help design and finance economically optimal fisheries enforcement systems.

1. Introduction

Illegal, unregulated, and unreported (IUU) fishing is a well-known global problem threatening the sustainability of both large and small-scale fisheries and the health of marine environments (e.g. [1–3]). The presence of IUU fishing substantially increases the uncertainty associated with estimating stock status and fishing mortality, which makes determining a sustainable harvest level challenging [4,5]. IUU fishing has been identified as a major factor contributing to the decline, and in some cases, collapse, of a number of fish stocks [6–8]. Other common problems associated with IUU fishing include: ecosystem impacts, economic losses for legal fishermen, and an increased incentive for others to overfish [9,10]. The first study to evaluate the worldwide extent of IUU fishing estimated global IUU fishing in 2003 to be between 11 and 26 million tonnes annually, valued between 10 and 23.5 billion dollars [4] – a substantial amount considering that the estimated global catch of marine capture fisheries in 2012 was 79.7 million tonnes [11]. Given the evolving nature and importance of IUU fishing and unsatisfied with an IUU estimate that was over ten years old, a 2015 FAO-led workshop proposed that the FAO should lead an initiative to determine a new estimate and update that estimate every 5–10 years [12]. A more recent study by Pauly and Zeller estimated global unreported catch, the difference between globally reported catch and a reconstructed global catch, to be 32 million tonnes in 2010, which includes not only illegal catch but also unreported artisanal and subsistence catch, recreational fisheries catch, discards, and bycatch [5]. Given the large scale of the problem, determining effective, feasible methods for eliminating IUU fishing should be considered a high priority.

The primary driver for IUU fishing is economic incentive [10,2]. A fisher who behaves illegally in hopes of financial gain is influenced by the expected costs and benefits of non-compliance [8,13]. An enforcement system, defined as the surveillance of compliance with regulations and the prosecution of those who do not comply with regulations [14,15], can help to decrease the expected benefits from illegal activity and deter such behaviors. The expected profitability of illegal fishing is a function of the enforcement system, and is inversely related to the enforcement effort and probability of detection, the probability of prosecution, and the cost of the penalty (measured in fines, the loss of future earnings due to revoked fishing privileges, etc.) [13]. Therefore, as any of these three aspects increase, the expected profitability of illegal fishing will decrease. The ability for an enforcement system to effectively deter IUU fishing in a particular fishery will also

---

* Corresponding author.
E-mail address: gmcdonald@bren.ucsb.edu (G. McDonald).

[1] These authors contributed equally to the research and development of this paper.

http://dx.doi.org/10.1016/j.marpol.2016.02.003
0308-597X/© 2016 Elsevier Ltd. All rights reserved.

depend on the status of the stock, social and economic conditions in the fishery, regulations being enforced, and spatial characteristics of the fishery [13,15–17]

Effective enforcement of a fishery management system has the potential to address the ecological, economic, and social implications of illegal fishing [11,18–21]. Effective enforcement can lead to improved management outcomes by reducing biomass uncertainty caused by unaccounted harvest, which can undermine management efforts. Together with proper management, it can also reduce other negative ecological impacts and assure some level of conservation. [21,22]. The economic and social benefits of reducing illegal fishing are likely to be substantial for developing nations where IUU fishing threatens both food security and livelihoods for those who depend on local fisheries as a protein source and means of income. Thus, effective enforcement can help to recapture dissipated fisheries benefits [14,23]. In the absence of IUU fishing, proper management of a stock will most efficiently maximize revenues [20]. Despite the clear benefits most fisheries would receive from improved enforcement, many fisheries lack an adequate level of enforcement, particularly in small-scale fisheries in developing countries [24,25].

Major barriers to more pervasive and effective enforcement include significant upfront capital costs and high operational costs of ongoing implementation [26,27]. Enforcement is generally the most expensive aspect of fishery management costs and increasing enforcement effort is often costly [15,13]. This is particularly a problem in small-scale fisheries in developing countries that depend on coastal fishing for livelihoods and food security, yet lack the resources to pay for enforcement [28,29]. Often, governments, NGOs, private investors, or a combination provide funding at the onset of new fisheries management initiatives but are unable to fund ongoing enforcement costs [30]. An enforcement system that is designed to eventually be self-financing not only ensures the sustainability of the fishery over time, but can also help to attract the upfront investments needed at the onset of enforcement reform. For a cost-recovery system in which the sectors benefiting from enforcement are responsible for financing this service, potential sources of funding for ongoing enforcement effort include license fees, taxes on landings, fines from illegal activity, and, if applicable, taxes on a relevant tourism industry such as diving. Cost-recovery has been used to finance the costs associated with fisheries management primarily in developed nations including the United States, Australia, New Zealand, Iceland, and Canada [15,31] but has also been used in developing nations including Uganda and Namibia [32,33]. In many cases, traditional funding sources (e.g., funding from local and national governments, foreign investment) are not available for fisheries enforcement. While cost-recovery programs are not yet widespread in small-scale fisheries of the developing tropics, this type of system has the potential to provide funding for fisheries enforcement and management activities in locations where other funding sources either do not exist or do not provide adequate resources for effective management.

The basic economic theory of fisheries enforcement has been previously developed and reported in the literature [34]. This theory posits that instituting a particular enforcement system in a fishery leads to a certain probability that fishers operating illegally will be apprehended and penalized. The probability of receiving a penalty is a function of the enforcement effort applied, the effectiveness of the particular enforcement method in terms of detecting violators, and the likelihood of prosecution. As profit maximizing individuals, illegal fishers take this information into account by including the expected penalty cost into their private benefit function and adjusting their fishing effort accordingly. It should be noted that this theory relies on the assumption that fishers are profit-maximizing, which may not always be the case. Other non-monetary factors may increase compliance including moral standards [26], social punishments for rule violators including ostracism and social degradation [35,36], fisher involvement with co-management [37], and fisher participation in cooperatives [38].

Stakeholder Archetypes

Archetype 1: Fishing industry only
Archetype 2: Dive tourism only
Archetype 3: Fishing and dive tourism industry

Fig. 1. Description of stakeholder archetypes.

Previous studies examined how optimal enforcement levels depend on stock status, the economic parameters of a fishery, and the enforcement system in place [39]. This optimal level was determined by maximizing the total social benefits in the fishery, accounting for both fishing profits as well as the cost of enforcement. This study investigates optimal enforcement levels using a bio-economic model in the context of an illustrative small-scale Caribbean lobster fishery in Barbuda and builds upon previous research in two important ways. First, the optimal enforcement effort level is determined as a function of the lobster stock status for three different stakeholder archetypes common in tropical small-scale fishery settings, each with private industries deriving benefits from the stock: (1) lobster fishing industry only; (2) dive tourism industry only; and (3) lobster fishing and dive tourism industries (Fig. 1). Second, the model is used to explore the potential of sustainably financing enforcement in a small-scale setting through financing mechanisms potentially available in small-scale fisheries: (1) fishing license fees; (2) a landings tax; (3) penalties received through the enforcement system; (4) a tax on dive tourism revenue. Therefore, the model is used to investigate the following questions: (1) how will optimal enforcement and fishing effort change given the stakeholder archetype and status of a stock; and (2) how can this optimal enforcement level be financed. This analysis may be used to help managers determine optimal enforcement levels for a fishery given the status of the stock and stakeholder archetypes, as well as to inform sustainable enforcement financing mechanisms and appropriate time-scales of recovering enforcement costs.

This paper will first describe the study site, an illustrative Caribbean lobster fishery in Barbuda. Next, the bio-economic model will be defined, which is composed of a biological model, dive tourism and fishery economic models, and an enforcement model. This section also includes a sensitivity analysis to the assumed starting lobster biomass and enforcement financing parameters. A description of how the model was parameterized for the study site is also provided. Next, modeling results are described that show how optimal enforcement and fishing effort depend on stakeholder archetype and stock status, as well as how this optimal enforcement can be financed. These results, along with a discussion of important assumptions and how these results are particularly important for the context of small-scale fisheries, are presented in the Sections 4 and 5.

2. Methods

2.1. Study site description

This study focuses on determining optimal enforcement levels and cost-recovery mechanisms for an illustrative Caribbean spiny lobster (*Panulirus argus*) fishery in Barbuda that includes both fishing and dive tourism industries. Barbuda is a small island (160.56 km^2) that makes up part of the Antigua- Barbuda State, located in the leeward islands of the eastern Caribbean. The lobster fishery is the most economically important fishery in Barbuda, with approximately 26% of Barbuda's population directly dependent on the lobster fishery as a source of income [40]. Until recently, the Antigua and Barbuda Defense Force (ABDF) was tasked with enforcing fishery regulations in Barbuda, which included trips to Barbuda from the headquarters in Antigua once or twice a month [41]. In 2012, compliance with the lobster fishery regulations was estimated to be 63.2% and increasing compliance and enforcement in the lobster fishery was identified as a priority to avoid growth overfishing [40]. The *2014 Barbuda Fisheries Regulations and The Barbuda Costal Zoning and Management Regulations* resulted in the establishment of five marine reserves and defined fishery fees, diving fees, and fishery violation fines.

2.2. Bio-economic model

For each stakeholder archetype in this small-scale fishery setting, a dynamically economically optimal legal harvest level for the lobster fishery is derived assuming no illegal fishing occurs. This optimal harvest level, which can be thought of as the lobster fishery's Total Allowable Catch (TAC), is a function of lobster stock biomass and maximizes industry-derived benefits. Once the optimal legal TAC for the fishery is determined, illegal fishing is introduced to determine an optimal enforcement level that maximizes a net social benefit function, which includes both industry benefits and enforcement costs. As part of this optimization, the level of illegal fishing that occurs each year is determined as a function of biomass and enforcement effort. Finally, the model is used to examine how and over what time scale the costs of enforcement can be financed through cost-recovery using the four financing mechanisms.

2.2.1. Biological model

A simple discrete logistic model [42] is used to describe population growth of the Caribbean spiny lobster *Panulirus argus* and calculate stock biomass (B) at an annual time step (t) after total annual catch (Q_t) from the fishery has been removed:

$$B_{t+1} = B_t + rB_t - \frac{rB_t^2}{K} - Q_t \tag{1}$$

Where r is the estimated intrinsic growth rate parameter and K is the lobster population's carrying capacity.

2.2.2. Economic model

A modification of the economic model described in [43] is used to calculate revenue generated from a fishing industry and dive tourism industry for the 3 stakeholder archetypes: (1) fishing industry only; (2) dive tourism industry only; and (3) dive and fishing industries. Revenue generated from both the fishing and dive tourism industries are assumed to be functions of lobster stock biomass (B).

2.2.2.1. Dive tourism model. The following equation described by Sala et al. [44] defines the relationship between the marginal value (P_t) of dives given lobster biomass:

$$P_t = \alpha_0 + \alpha_1 D_t + \alpha_2 B_t \tag{2}$$

Where D_t is the number of dives occurring in each year t, B_t is the lobster biomass in each year, α_0 is a dive tourism value parameter that represents the value of the first dive even without any lobster biomass in the water, α_1 is a dive tourism value parameter that reflects the assumption that additional dives (after the first dive) are marginally less valuable, and α_2 is a dive tourism value parameter that describes the assumed positive linear relationship between dive value and lobster biomass. The number of dives that maximizes dive tourism revenue is then described as follows:

$$D_t = \frac{\alpha_0 + \alpha_2 B_t}{-2\alpha_1} \tag{3}$$

Using Eqs. (2) and (3), the optimal price per dive is determined as:

$$u_t = \alpha_0 + \left(\frac{\alpha_0 + \alpha_2 B_t}{-2}\right) + \alpha_2 B_t \tag{4}$$

And finally, the total revenue generated from the dive tourism industry in archetypes 2 and 3 is calculated as:

$$R_t = D_t u_t \tag{5}$$

By including this relationship in the profit functions for archetypes 2 and 3, profits are generated not only by harvesting fish out of the water, but also by non-extractive activities when some fish are left in the water. This changes the dynamic for determining optimal catch levels and enforcement effort.

2.2.2.2. Fishing industry. The model is used to determine a dynamic optimal legal harvest level (or TAC, denoted by Q_t^*) for the lobster fishing industry and the corresponding actual harvest response that includes illegal fishing (Q_t) using a 2-step optimization process. For archetype 2, which assumes only a dive industry is present, $Q_t^* = 0$ because no legal fishery exists for this archetype. For archetypes 1 and 3, Q_t^* is determined by maximizing the present value of a private industry benefit function that includes the fishing industry, the dive tourism industry, or both (π^*, Eqs. (6) and (7) below). These equations are used to find the optimal TAC, as a function of lobster biomass, that maximizes net present value (NPV) of the fishing and/or diving industries present assuming that no illegal fishing occurs. The private lobster fisher benefit function for archetype 1 ($\pi_{t,1}^*$) is defined as:

$$\pi_{t,1}^* = pQ_t^* - c\frac{Q_t^{*2}}{B_t} - L_t - v_t Q_t^* \tag{6}$$

Where, p is the price paid per unit weight of total legal lobster catch (TAC) (USD/kg), c represents the cost of lobster fishing (USD/kg), L is the cost of fishing licenses for the entire lobster fishing fleet (USD), and v is the tax paid per unit weight of catch of lobster (USD/kg).

The private combined lobster fishing and dive tourism benefit function for archetype 3 ($\pi_{t,3}^*$) is defined as:

$$\pi_{t,3}^* = pQ_t^* - c\frac{Q_t^{*2}}{B_t} - L_t - v_t Q_t^* + (1 - w_t)R_t \tag{7}$$

Where w_t is an assumed tax placed on the dive tourism industry revenue.

Given the optimal TAC calculated above (Q_t^*), the actual total annual catch response (Q_t) landed by both legal and illegal fishing is determined by maximizing a modified private industry benefit function (π) that allows for illegal fishing to occur ($Q_t > Q_t^*$). The private industry benefit function used to determine Q_t for each archetype are defined as:

$$\pi_{t,1} = pQ_t - c\frac{{Q_t}^2}{B_t} - f\phi(e_t)(Q_t - Q_t^*) - L_t - v_t Q_t^* \tag{8}$$

$$\pi_{t,2} = (1 - w_t)R_t \tag{9}$$

$$\pi_{t,3} = pQ_t - c\frac{Q_t^{\,2}}{B_t} - f\phi(e_t)(Q_t - Q_t^*) - L_t - v_t Q_t^* + (1 - w_t)R_t \tag{10}$$

Where, $f\phi(e_t)(Q_t - Q_t^*)$ represents the total expected fine illegal fishers will be levied (USD) given a fine parameter f (USD/kg), the probability of receiving a fine (ϕ) given the enforcement effort level (e), and the amount of landings that exceed the legal harvest level. Note that while Eq. (9) only includes dive tourism profit due to the definition of this archetype, illegal fishing will still occur when it is profitable to do so (thus illegal catch Q_t for archetype 2 will be $[p - f\phi(e_t)B_t]/(2c)$ anytime this number is positive, and zero otherwise). The method for determining the enforcement level e is discussed in Section 2.2.3.2. For all archetypes, the actual catch calculated each year (Q_t) is incorporated into the biological portion of the model to calculate the lobster biomass (B_t) for the following year (Eq. (1)).

2.2.3. Enforcement model

2.2.3.1. Enforcement parameters. The probability of an illegal fisher being detected, apprehended, prosecuted, and fined (ϕ) is assumed to be a function of the enforcement effort level (e_t), which can range from 0 to 1. There are several functional forms that could be used to describe this relationship, although for this study the following form is used [43]:

$$\phi_t(e_t) = \begin{cases} e_t;\, 0 < e_t \le 0.1 \\ a + b\ln(e_t);\, 0.1 < e_t \le 1 \end{cases} \tag{11}$$

where a represents the detection probability at the maximum enforcement effort level (which occurs when $e = 1$). The natural log of enforcement effort is used to change the shape of the curve to represent diminishing returns with marginal increases in enforcement effort – this is an assumption the authors make and consistent with previous studies [43]. Since the natural log of 0 is undefined, ϕ was assumed to be linearly proportional to e_t for effort levels between 0 and 0.1.

The cost of enforcement (C) given the enforcement effort level (e) at each time step (t) is defined as:

$$C_t(e_t) = C_0\{t = 0\}$$
$$C_t(e_t) = n_{pb} e_t h_{max}\left(C_{fuel} + n_{po} C_{po}\right)\{t >= 1\} \tag{12}$$

Where C_0 represents fixed enforcement costs associated with the investment in enforcement materials. C_t for $t \ge 1$ describes annual variable costs, where n_{pb} is the number of patrol boats, e is the enforcement level ranging from 0 (no enforcement) to 1 (maximum enforcement), h_{max} represents the maximum number of patrol vessel hours in a year, C_{fuel} is the hourly fuel cost of operating one patrol boat, n_{po} is the number of patrol officers per patrol boat, and C_{po} is the hourly rate per patrol officer.

2.2.3.2. Determining optimal enforcement effort and financing revenue streams. To determine the optimal enforcement effort for each archetype, net social benefit functions are first defined for each archetype ($x_{t,1}$ $x_{t,2}$ and $x_{t,3}$) that include: the private industry benefit(s) for that particular archetype (π, defined in Eqs. (8), (9), and (10) above); the costs of enforcement given enforcement effort ($C(e)$); and financing revenues generated by a fishing license fee, a landings tax, penalties received through the enforcement system, and a tax on dive tourism revenue. Note in Eq. (14) that since no legal fishery exists for archetype 2 by definition, any catch is illegal and subject to being fined if detected.

$$x_{t,1} = \pi_{t,1} - C_t(e_t) + f\phi(e_t)(Q_t - Q_t^*) + L_t + v_t Q_t^* \tag{13}$$

$$x_{t,2} = \pi_{t,2} - C_t(e_t) + f\phi(e_t)(Q_t) + w_t R_t \tag{14}$$

$$x_{t,3} = \pi_{t,3} - C_t(e_t) + f\phi(e_t)(Q_t - Q_t^*) + L_t + v_t Q_t^* + w_t R_t \tag{15}$$

For each archetype, the dynamic optimal enforcement effort (e) is determined by finding the enforcement effort as a function of biomass that maximizes the NPV of social benefit x over a time period of a 20-year time horizon and a discount rate of 0.05. Twenty years was chosen in order to allow the system to reach equilibrium conditions, while it is also assumed to be short enough to represent a meaningful time horizon for the social planners of this illustrative Caribbean lobster fishery. The total costs of enforcement alongside the financing revenue streams are also found at each time step to determine at what time (if any) social planner revenue exceeds enforcement costs. This is later referred to this as the social planner break-even point.

2.2.3.3. Determining enforcement effort necessary to eliminate illegal fishing. An additional analysis is conducted to determine the enforcement effort, as a function of lobster biomass, necessary to completely eliminate illegal lobster fishing and thus perfectly achieve the legal harvest level (TAC) for each archetype. These enforcement levels are allowed to be higher than the optimal levels. For each archetype, the additional cost needed to achieve this enforcement effort is also found. Although this enforcement effort is thus not economically optimal, it may be more desirable from a social or conservation perspective.

2.3. Sensitivity analyses

2.3.1. Impacts of initial stock status

The impact of starting stock status (B/B_{MSY} at $t=1$) on optimal enforcement effort levels and costs, social planner break-even point, financing revenue streams, and net social benefits is investigated by running the bio-economic model and enforcement model for initial stock levels of B/B_{MSY}=0.4, 0.8, 1.2, 1.6, and 2.0 (Table 1). This is done for each of the three archetypes.

2.3.2. Impact of financing mechanism parameters

The sensitivity of the results to the landings tax, enforcement fine, tourism revenue tax, and fishing license fee parameter values is investigated. Each parameter is run over the range of values specified in Table 1 while all other parameters are held at their base value.

2.4. Model parameterization

Parameters were chosen that are representative of the lobster fishery of Barbuda (Table 2). In some cases, parameters are drawn directly from the Barbuda lobster fishery. In other cases, when data were not directly available, reasonable estimates were chosen that are representative of conditions in a small-scale fishery based on previously published data of other small-scale fisheries, personal observations, and personal communications with local experts [41].

Table 1
Sensitivity analysis of input parameters, including the values used in the base model and the range of values examined to determine the model's sensitivity to each parameter.

Parameter	Base value	Range examined
Starting stock biomass (B_0/B_{MSY})	0.5	0.4–2
Landings tax (% of total landings value)	5	2.5–25
Cost of fishing license ($/year)	18	0–895
Illegal fishing fine (scalar to landings price)	100	25–250
Tourism tax (% of total dive tourism revenue)	5	2.5–25

Table 2
Fixed model parameters.

	Parameter	Value	Units	Description	Reference
Biological	r	0.798	–	Intrinsic growth rate	[45].
	K	1.29 E+06	kg	Carrying capacity	Estimated; See S.I.
Economics	c	7	USD/ kg	Cost of fishing per kg	Personal communication with Barbuda Fisheries Division
	p	10	USD/kg	Price for kg of landed lobster	Personal communication with Barbuda Fisheries Division
	d	0.05	–	Discount rate	Assumed
Tourism	α_0	9.64	USD	Tourism parameter 0	[44].
	α_1	−0.003	USD/dive	Tourism parameter 1	[44].
	α_2	0.000122	USD/kg	Tourism parameter 2	Estimated; See S.I.
Enforcement	a	0.25	–	Probability of receiving a fine parameter	Estimated; See S.I.
	b	0.1026	–	Probability of receiving a fine parameter	[43].
	$C_t(e_t)\ \{t=1\}$	60,000	USD	Fixed cost of enforcement at time 0	Personal communication with Barbuda Fisheries Division
	n_{pb}	3	Boats	Number of patrol boats	Assumed
	h_{max}	2920	Hours	Maximum number of patrol vessel hours in a year	Assumed (365 days/year at 8 hours/day)
	n_{po}	2	Officers/boat	Number of patrol officers per boat	Assumed
	C_{fuel}	2.91	USD/hour/boat	Cost of fuel per patrol boat per hour	Personal communication with Barbuda Fisheries Division
	C_{po}	30	USD/officer/hour	Salary per patrol officer per hour	Personal communication with Barbuda Fisheries Division

2.4.1. *Biological model*

The population's intrinsic growth rate parameter (r) was borrowed from the Jamaican spiny lobster population [45]. Information from the Barbuda spiny lobster fishery was used to estimate the carrying capacity (K). At the time of the last assessment in 2013, it was determined that the Barbuda lobster fishery was being harvested at a sustainable level, [40]. Thus, annual landings were assumed to equal *MSY*. Data on total annual landings in the Barbuda lobster fishery were not available, but an approximation of total annual landings was calculated using known fishery parameters from [40] (Table 3):

$$MSY = \text{number of fishing vessels} * \text{average trips per vessel per year} * \text{average catch per trip} \tag{16}$$

K was then calculated using the relationship:

$$K = \frac{4MSY}{r} \tag{17}$$

2.4.2. *Economic model*

Model parameters used to define the costs and revenues generated from lobster fishing were reasonably assumed based on personal communication with local experts.

2.4.3. *Dive tourism model*

Although no data currently exists on the relationship between fish (or lobster) biomass in the water and dive frequentation, 50% of all dives in the Caribbean take place within a marine reserve, indicating divers' preference to frequent areas with more abundant marine life [47]. Currently there is little to no dive tourism in Barbuda. However, the recent establishment of five marine reserves in Barbuda waters may increase the island's attraction to divers and present an opportunity expand the tourism dive industry. Diver preference parameters α_0 and α_1 (Eqs. (2), (3), and (4)) are taken from the Medes Islands Marine Reserve in Spain [44]. α_2 is estimated by assuming that the maximum possible tourism revenue (generated when the stock is at carrying capacity K) should be equal to the maximum sustainable fishery revenue (generated when the legal catch is at *MSY*). By making this assumption, the fishing and tourism industries are normalized in order to draw more intuitive results from model simulations.

Table 3
Information from the literature used to derive the carrying capacity (K).

Parameter	Value	Units	Description	Reference
Number of legal fishing vessels	34	Vessels	–	[40].
Average landings per trip	63	kg/trip	–	[40].
Average trips per year per vessel	120	trips	–	[46].

2.4.4. *Enforcement model*

The number of patrol boats (n_{pb}), number of patrol officers per boat (n_{po}), and maximum number of patrol vessel hours in a year (h_{max}) were assumed to be three, two, and 2920 respectively (Table 2). The cost of fuel per patrol boat per hour (C_{fuel}) and the salary per patrol officer per hour (C_{po}) are based on personal communication with the Barbuda Fisheries Division and published values [41] (Table 2). These values were used to determine the variable enforcement cost using Eq. (12).

To determine the relationship between enforcement effort and the probability of detecting illegal fishing (Eq. (11)), it is assumed that the probability is proportional to the amount of area patrol vessels are able to cover in a day relative to the total fishing area. Barbuda's waters include an estimated total fishing area of 886 km^2 [46]. Patrol vessels were assumed to patrol for 8 h a day at an average speed of 15 knots, and each cover a 27 km swath (13.5 m line of vision on either side of the vessel). Given these assumptions, one patrol vessel could cover a total area of 216 km^2 over 8 h, or approximately 25% of the total fishing grounds. Assuming 3 patrol vessels patrolling for 8 h a day, 100% enforcement effort corresponds to covering 75% of the total fishing area. Assuming fishers operate 24 h per day, a is calculated to be 0.25

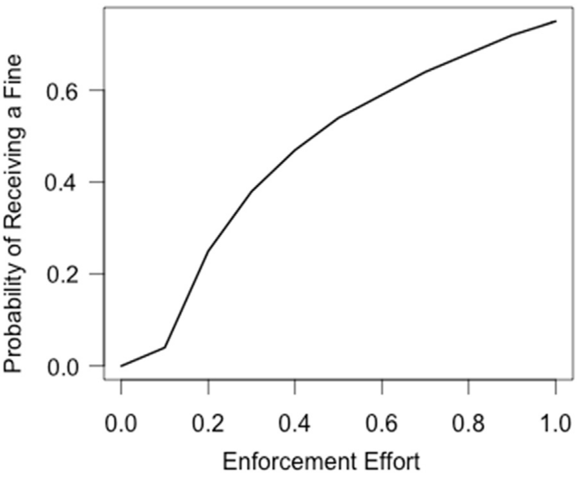

Fig. 2. Probability of receiving a fine as a function of enforcement effort for the Caribbean spiny lobster fishery.

(such that when enforcement effort is equal to 1, the detection probability is 25%). Currently, no data exists from a small-scale fishery the Caribbean to determine the shape parameter *b*, thus, this value is borrowed from the enforcement detectability of patrol vessels in the Kattegat and Skagerrak nephrops fishery [43]. Using Eq. (11) and these parameters, the relationship is shown in Fig. 2.

3. Results

3.1. The impact of biomass and archetype on optimal effort levels, optimal enforcement levels, and illegal fishing effort

For each stakeholder archetype (Fig. 1), the model finds the optimal annual fishing effort and dynamic optimal enforcement effort as functions of stock biomass (B/B_{MSY}) (Fig. 3a and b). The optimal fishing effort (and TAC) increases as lobster biomass increases for archetypes 1 and 3 – there is no legal fishing effort for archetype 2, which only has dive tourism (Fig. 3a). For archetype 1, the optimal fishing effort is zero when lobster stock biomass is below $B/B_{MSY}=0.6$ and then increases with increasing biomass. Similarly, the optimal fishing effort for archetype 3 is zero when lobster biomass is below $B/B_{MSY}=1.0$ and then increases with increasing biomass. Above $B/B_{MSY}=0.6$, optimal fishing effort for archetype 1 is always greater than for archetype 3.

For all three stakeholder archetypes, optimal enforcement effort is highest when lobster biomass is low, and decreases as biomass increases (Fig. 3b). Optimal enforcement effort is always the highest in archetype 2, which consists of only a dive tourism industry. At low biomass levels, the optimal enforcement levels for archetypes 1 and 3 are the same. At $B/B_{MSY}=0.1$, enforcement effort for archetypes 1 and 3 is about 0.23 while enforcement effort for archetype 2 is only slightly higher at about 0.24 (corresponding to roughly 1.82 and 1.91 h of surveillance per day, respectively). However, at higher biomasses ($B/B_{MSY}>0.6$), optimal enforcement levels are higher for archetype 3 than archetype 1. When $B/B_{MSY}=0.8$, enforcement effort is equal to 0.2, 0.24, and 0.23 for

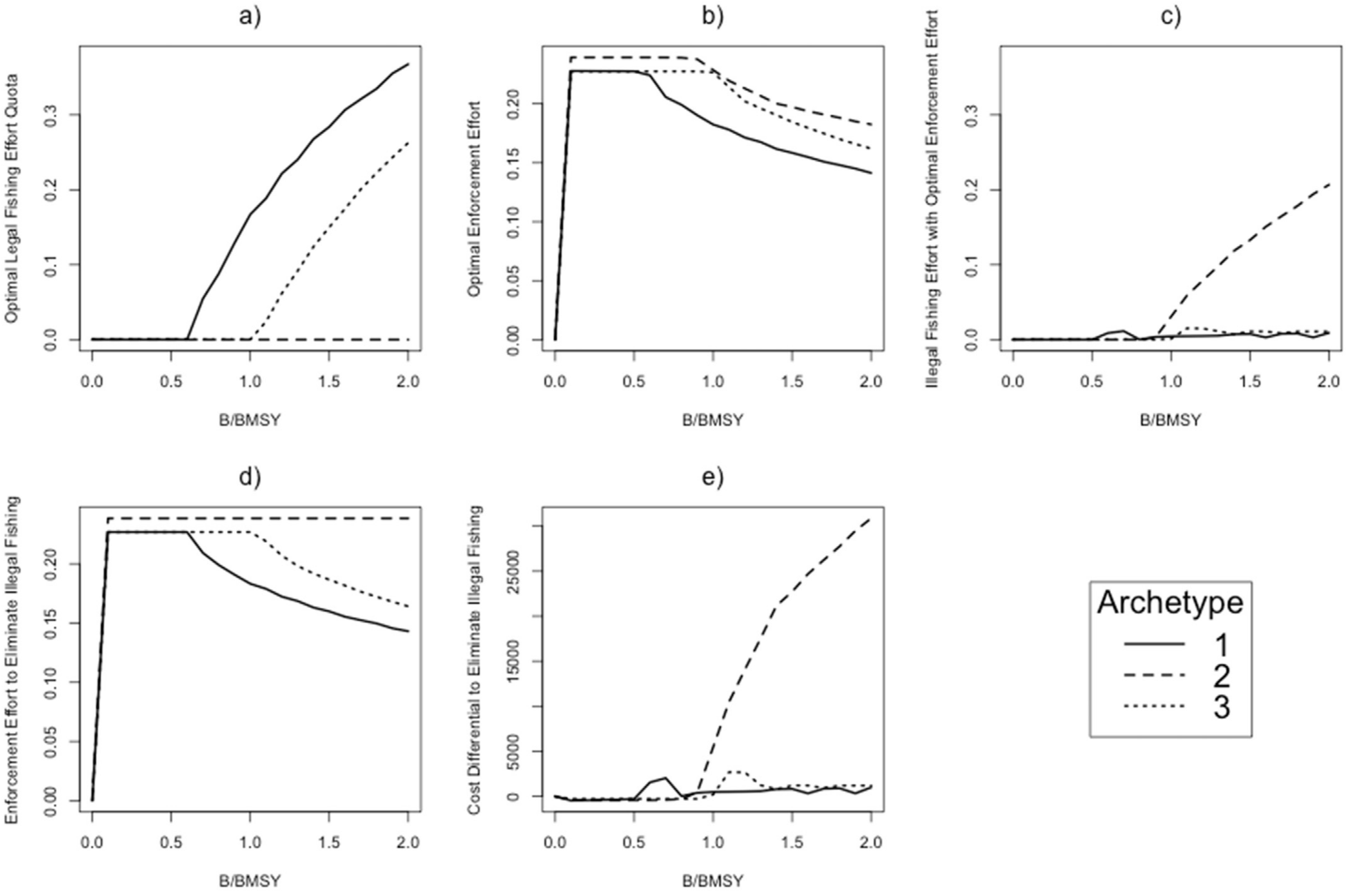

Fig. 3. (a) Optimal fishing effort quota given stock status B/BMSY; (b) Optimal enforcement effort given stock status; and (c) illegal fishing effort response given stock status; (d) enforcement effort necessary to eliminate illegal fishing; and (e) cost differential between optimal enforcement effort and enforcement effort necessary to eliminate illegal fishing. The stakeholder archetypes are defined as (1) fishing only; (2) dive tourism only; and (3) fishing and dive tourism.

archetypes 1, 2, and 3 respectively (corresponding to roughly 1.59, 1.91, and 1.82 hours of surveillance per day, respectively). Thus when biomass is high enough, optimal enforcement levels are highest when both fishing and dive tourism are present. Notably, optimal enforcement efforts are always substantially lower than 1 regardless of archetype – therefore it is never optimal to employ 100% surveillance.

The model is also used to determine how the optimal fishing effort and optimal enforcement effort affect illegal fishing effort (Fig. 3c). For all three archetypes, illegal fishing effort is influenced by archetype, biomass level, and enforcement effort. Illegal fishing effort increases as the biomass increases, reflecting increased profitability of illegal fishing when more fish are in the water. For low biomass levels ($B/B_{MSY} \leq$ to 0.5), illegal fishing effort for all three archetypes is zero, reflecting the high enforcement levels for these conditions. For higher biomass levels, illegal fishing effort is highest for archetype 2, followed by 3 and 1 respectively, which have relatively minimal illegal fishing regardless of stock status. Archetype 2, which by definition has a legal fishing effort of zero, has significant illegal fishing above $B/B_{MSY}=1$.

Finally, the model is used to determine the additional enforcement effort and associated enforcement cost necessary to fully eliminate illegal fishing for each archetype (Fig. 3d and e). At stock conditions below $B/B_{MSY}=0.5$, the economically optimal enforcement effort already eliminates illegal fishing, so there is no need for additional enforcement. Above this biomass, however, the enforcement effort necessary to eliminate illegal fishing is higher than the economically optimal enforcement effort for all three archetypes. The additional enforcement effort necessary to eliminate illegal fishing at $B/B_{MSY}=1.2$ is 0.001, 0.025, and 0.005 for archetypes 1, 2, and 3 respectively. The corresponding additional annual enforcement cost is $531, $14,045, and $2664 for the archetypes 1, 2, and 3 respectively. This additional enforcement effort, and associated cost, is highest for archetype 2. Since by definition there is no legal fishing allowed for this archetype at all, this archetype requires the highest enforcement effort to fully disincentivize illegal fishing.

3.2. Financing enforcement

While the above results demonstrate that in many cases it is economically optimal to monitor and enforce a fishery, financing that enforcement effort is a perennial challenge for many of the world's fisheries. There are many mechanisms through which the beneficiaries of enforcement (i.e., legal fishers or divers) could be used to help finance enforcement – a small set of them is examined here. The four social planner revenue streams analyzed in this study include the following: (1) a license fee ($18/year/fisher); (2) a landings tax of 5%; (3) fines from illegal fishing ($100 per kg of illegal harvest, or 10 times the legal ex-vessel price); and (4) a tax on dive tourism revenues of 5%. These values were chosen arbitrarily, and sensitivity analysis for each of the four financing parameters was conducted. When projecting the expected enforcement costs and social planner revenue streams over time, it is found that these values differ based on the starting stock biomass and the stakeholder archetype. For all three archetypes, higher starting stock biomasses lead to lower NPVs of enforcement costs and higher NPVs of financing revenues over a twenty-year time horizon (Fig. 4). For archetypes 1 and 3, the financing stream that generates the most revenue for the social planner is the landings tax. For archetype 2, fines generated by enforcement represent the highest fraction of social planner revenue, which reflects the fact that all fishing activities are illegal for archetype 2. Revenue generated by a fishing license fee was negligible compared to other financing streams for archetypes 1 and 3. Importantly, it is found that the NPV of revenues exceeds the NPV of enforcement costs for every archetype and starting stock condition, indicating that this suite of financing mechanisms can fully finance optimal levels of enforcement regardless of the industries present and starting stock conditions.

However, while the NPV of social planner revenues exceeds the

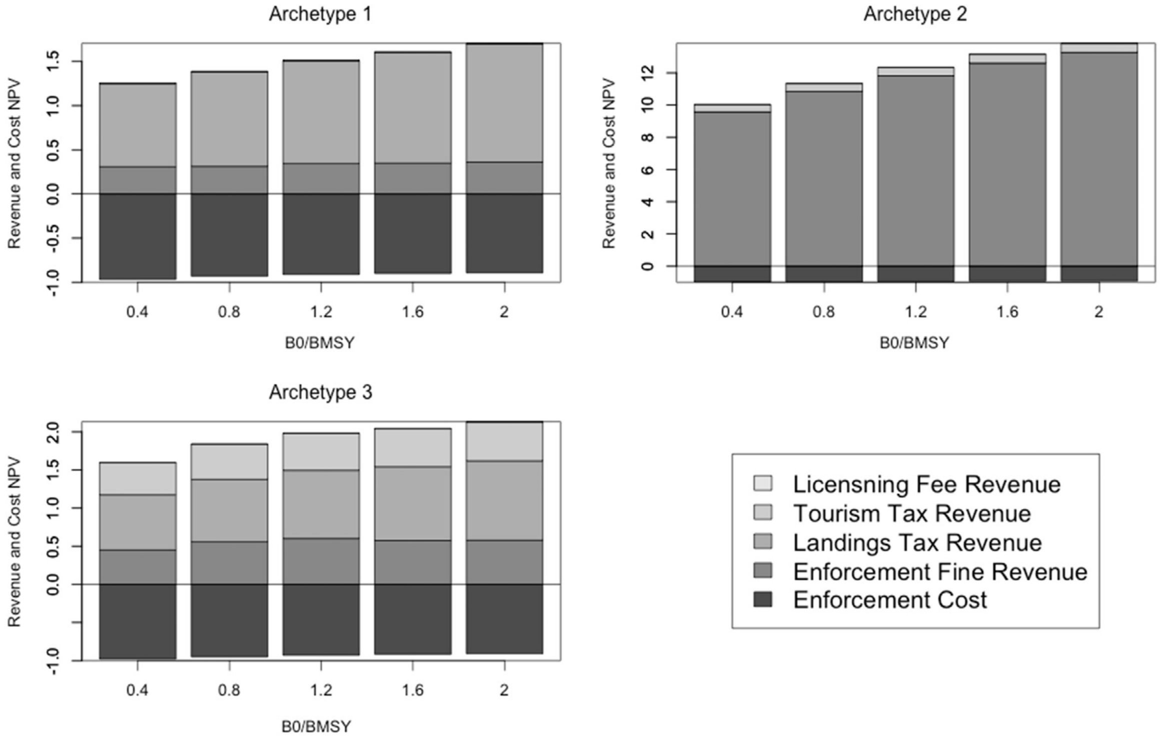

Fig. 4. Net present value (NPV) of social planner cost and financing revenue streams over a 20-year time horizon for various starting stock statuses (values are scaled to the NPV of the maximum enforcement cost for that particular archetype). The stakeholder archetypes are defined as (1) fishing only; (2) dive tourism only; and (3) fishing and dive tourism.

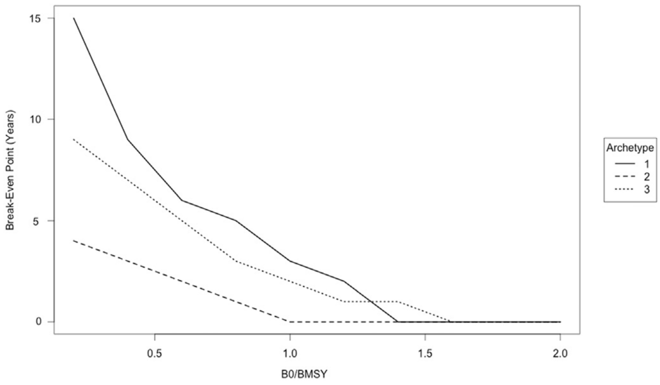

Fig. 5. Break-even point for each archetype. The stakeholder archetypes are defined as (1) fishing only; (2) dive tourism only; and (3) fishing and dive tourism.

NPV of enforcement costs for every scenario, the social planner break-even point (the year in which the NPV of social planner revenue first exceeds the NPV of enforcement costs up to that point) varied depending on the archetype and starting stock biomass (Fig. 5). Higher starting stock biomasses lead to shorter amounts of time before reaching break-even points for all archetypes. However, if the starting stock biomass is at least equal to 1.4B_{MSY}, 1.0B_{MSY}, and 1.6B_{MSY} for archetypes 1, 2, and 3 respectively, the break-even point occurs in the first year. This indicates that for these starting conditions, optimal levels of enforcement can be contemporaneously financed using this suite of financing mechanisms. In cases with a break-even point of one year or greater, external financing is needed to cover the initial fixed enforcement costs.

Sensitivity analyses indicate the dependency of these results on the various financing parameters that can be adjusted by the social planner (Figs. 6–8). These figures demonstrate the tradeoffs that social planners must consider when designing their financing mechanisms. The four parameters not only impact the industry NPV, but also the social planner break-even point for financing the cost of enforcement. By increasing the landings tax in archetypes 1 and 3, the social planner decreases the industry NPV but also shortens the time it takes to reach the break-even point for low starting biomass conditions. However, increasing the landings tax above 15% has no impact on the break-even point while continuing to negatively impact industry NPV. Increasing the enforcement fine for all 3 archetypes increases the industry NPV while also decreasing the time it takes to reach the break-even point, leading to win-win policy. However, above a certain enforcement fine (20 times the landings price), there are negligible impacts on industry NPV. In addition, there are negligible impacts on the social planner break-even point when the enforcement fine is between 10 and 15 times the landings price in archetype 1 (depending on the starting stock biomass) and about 15 times the landings price in archetypes 2 and 3. In archetypes 2 and 3, increasing the dive tourism tax decreases industry NPV but also slightly shortens the time it takes to reach the break-even point. In archetypes 1 and 3, increasing the fisher license fee slightly decreases the industry NPV while shortening the time it takes to reach the break-even point for biomass levels $\leq 1.2\ B_{MSY}$.

4. Discussion

4.1. The impact of biomass and archetype on optimal legal catch levels, optimal enforcement levels, and illegal fishing effort

As expected, the optimal TAC for the small-scale lobster fishery example is dependent on the lobster stock biomass and the

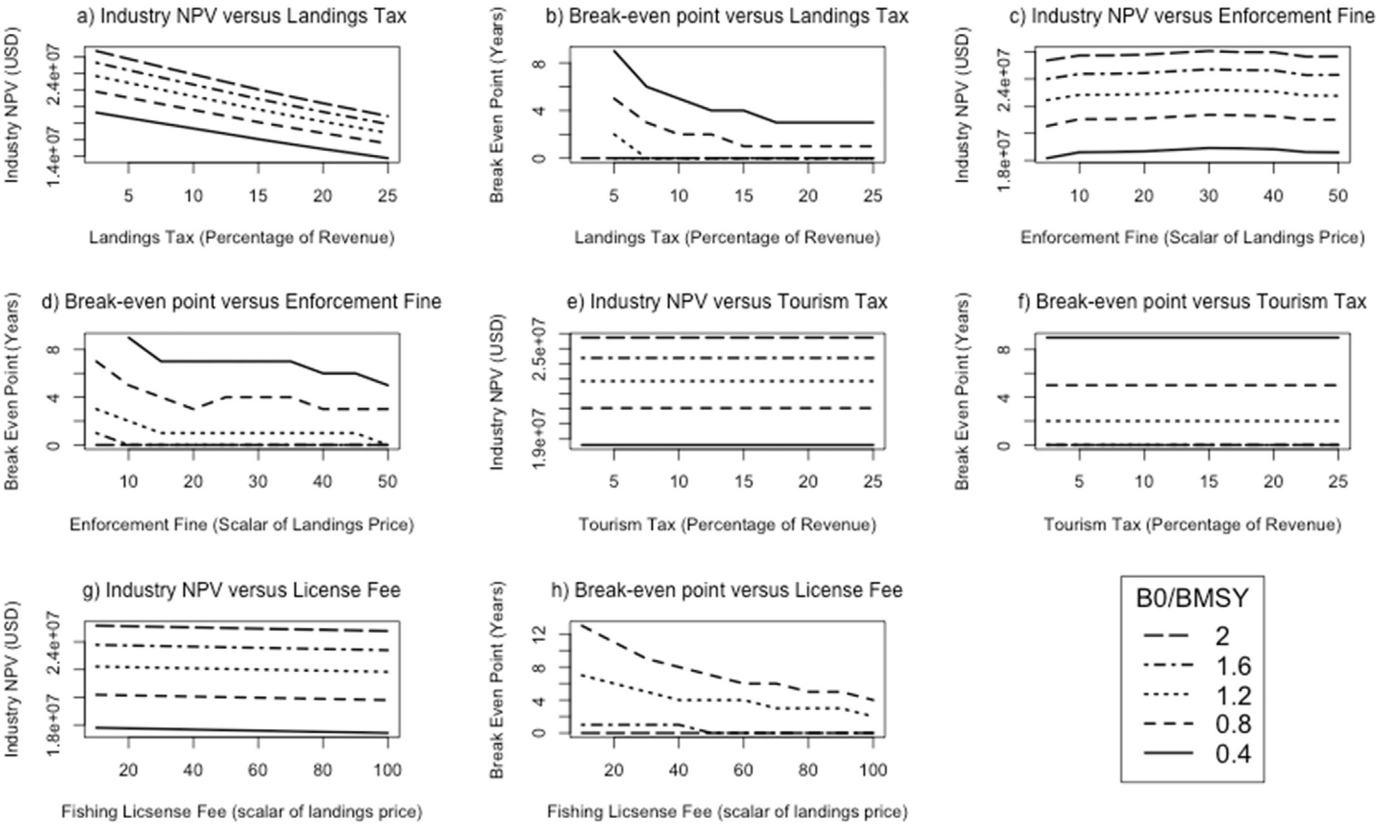

Fig. 6. Archetype 1 sensitivity analysis for financing mechanisms (fishing industry only). (a) Industry NPV versus Landings Tax; (b) Break-even point versus Landings Tax; (c) Industry NPV versus Enforcement Fine; (d) Break-even point versus Enforcement Fine; (e) Industry NPV versus Tourism Tax; (f) Break-even point versus Tourism Tax; (g) Industry NPV versus License Fee; and (h) Break-even point versus License Fee.

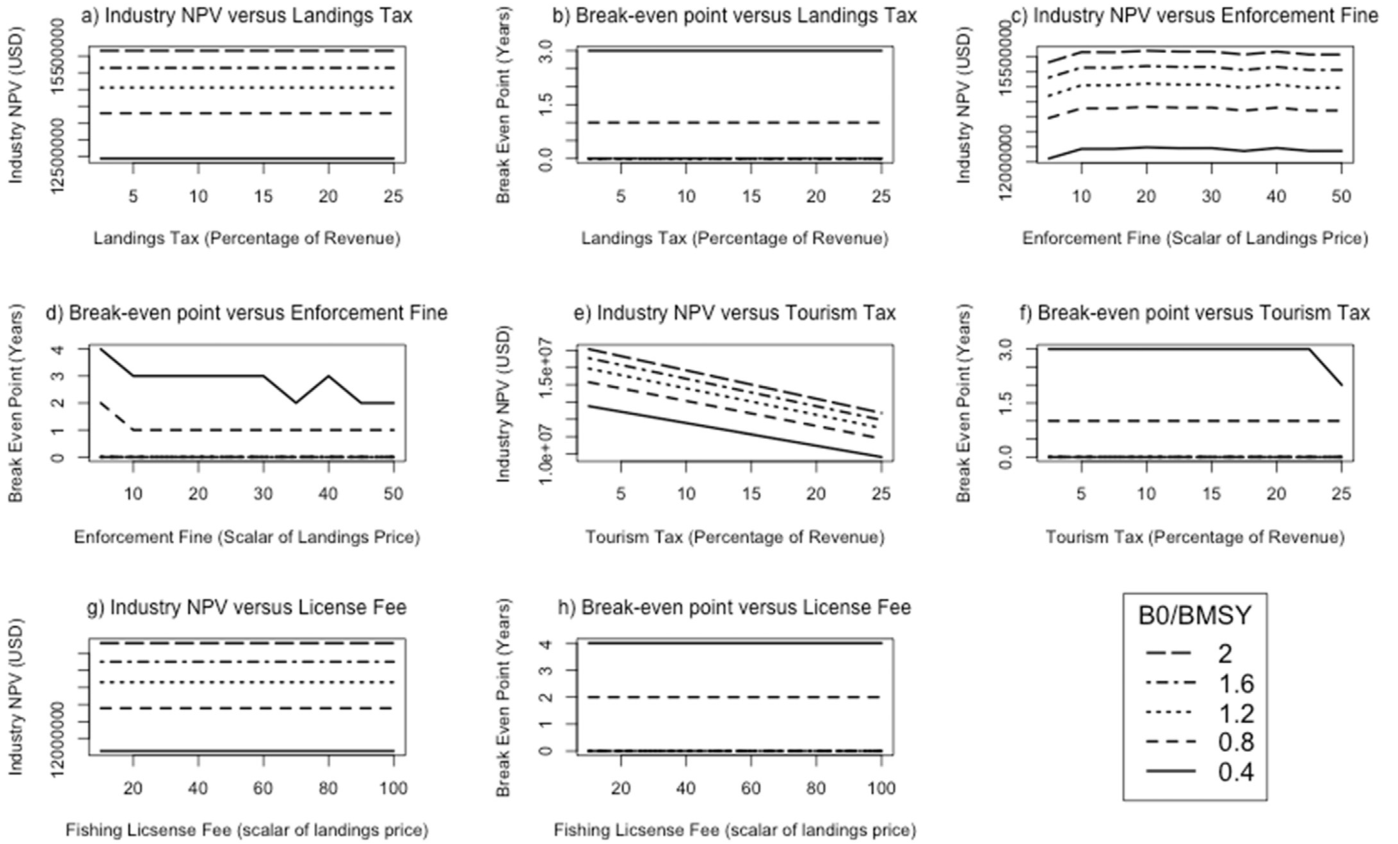

Fig. 7. Archetype 2 sensitivity analysis for financing mechanisms (tourism industry only). (a) Industry NPV versus Landings Tax; (b) Break-even point versus Landings Tax; (c) Industry NPV versus Enforcement Fine; (d) Break-even point versus Enforcement Fine; (e) Industry NPV versus Tourism Tax; (f) Break-even point versus Tourism Tax; (g) Industry NPV versus License Fee; and (h) Break-even point versus License Fee.

stakeholder archetype. For all stakeholder archetypes, the optimal TAC increases with increasing lobster stock biomass. The optimal TAC for the lobster fishery was always lower when a dive tourism industry was present, which makes intuitive sense; in this case, the value that the tourism industry receives from lobster biomass in the water is sufficient to reduce the economically optimal TAC and increase enforcement effort. Additionally, it makes intuitive sense that the optimal level of enforcement is not constant over time but rather depends on the stock status of the target fishery and/or species of interest for the dive tourism industry. At low levels of lobster stock biomass, the potential improvements to the stock and the net present value of benefits make it worthwhile to spend more money on costly enforcement. In other words, the shadow value of the biomass of an overfished stock is higher than that of biomass from a healthy stock in our lobster fishery example. At higher stock biomass levels, increases in stock biomass are smaller and thus it is more efficient to spend less money on costly enforcement, which may only yield a small increase in stock biomass. This result is consistent with previous findings in the literature [39]. This result has practical implications for managers who hope to implement an economically efficient enforcement system, especially in fisheries that are currently overfished; enforcement effort should be highest when the stock biomass is lowest (i.e. overfished), and can decrease over time as the status of the stock improves. This scenario may require a higher initial investment for enforcement costs, but lower ongoing costs. This may have important consequences for small-scale fisheries, which may be overfished and require these higher initial levels of enforcement. If local institutions are unable to pay for these initial costs, outside investment may be needed.

While this analysis only includes dive tourism and fishing industries, it is likely that including other stakeholders and ecosystem services in the analysis would alter optimal legal harvest levels; specifically, including other stakeholders who value biomass in the water would likely further reduce the optimal legal harvest levels and increase optimal enforcement effort [44,48]. For the purposes of this paper, the focus was placed on extractive and non-extractive direct use values generated by a fishing industry and dive tourism industry, respectively. However, there may be additional non-use values, such as the intrinsic existence value of a healthy ecosystem and conservation value [49–51]. Given the large numbers of conservation-focused NGOs working in the Caribbean as well as other regions in the developing tropics, including the Waitt Institute and WildAid in Barbuda [41], these non-use values could be important to include in future studies, especially since these values can be considerable [50,52]. Importantly, while there are a number of non-market valuation methods for quantifying these types of non-use values, these approaches are data-hungry and associated with strong assumptions and caveats [53,54]. However, even without explicitly including non-use values in this analysis, the example of dive tourism is demonstrative of how placing economic value on biomass in the water for whatever reason will likely decrease the optimal legal fishing quota and increase the optimal enforcement effort.

4.2. Financing enforcement

Improved enforcement of a sustainable fishery management plan can increase industry benefits, but is often associated with high costs [14,55,56]. National enforcement costs have been estimated to be between 0.4–37.2% of total fishery value [14]. However, as the model demonstrates, by establishing an effective sustainable financing plan,

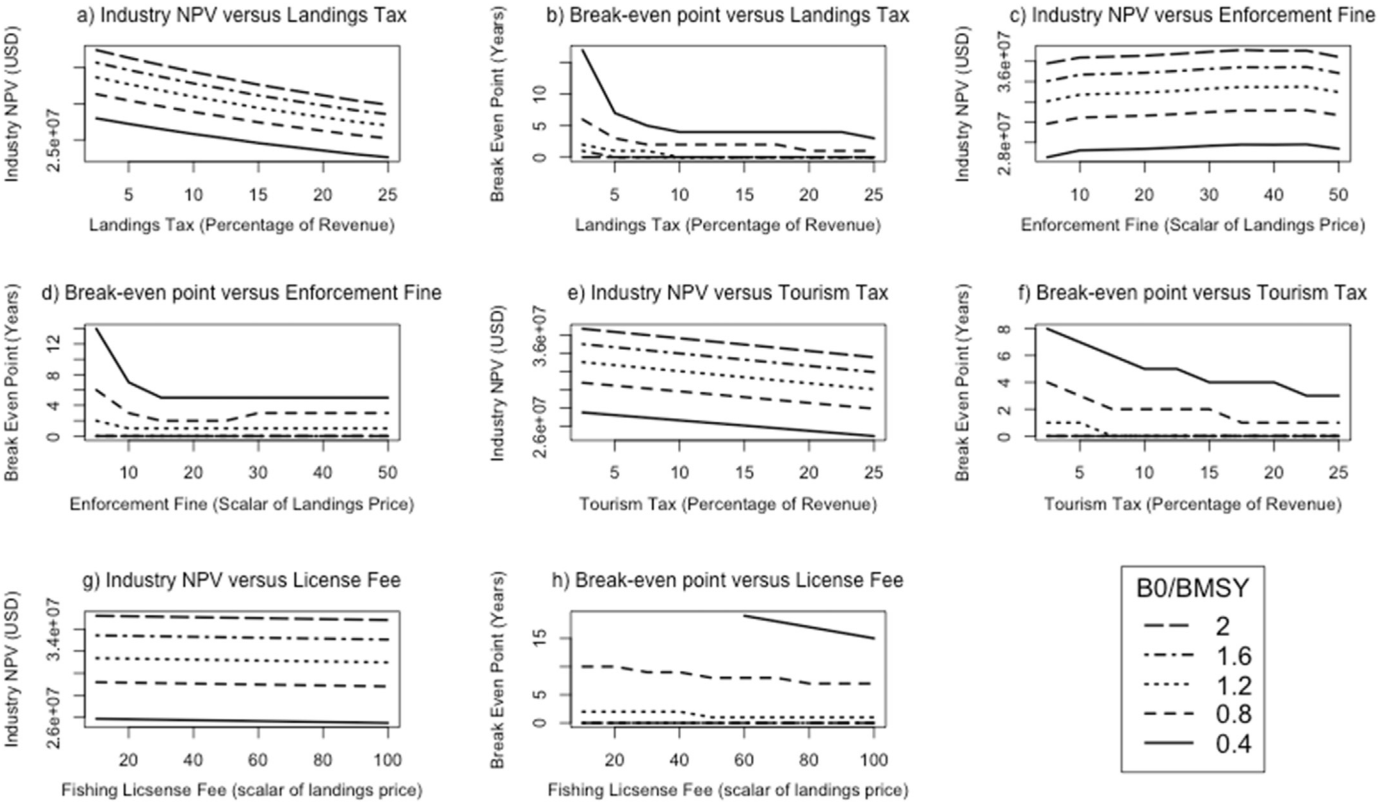

Fig. 8. Archetype 3 sensitivity analysis for financing mechanisms (fishing and tourism industries). (a) Industry NPV versus Landings Tax; (b) Break-even point versus Landings Tax; (c) Industry NPV versus Enforcement Fine; (d) Break-even point versus Enforcement Fine; (e) Industry NPV versus Tourism Tax; (f) Break-even point versus Tourism Tax; (g) Industry NPV versus License Fee; and (h) Break-even point versus License Fee.

even when there is no dive tourism industry, the cost of enforcement can be sustainably financed. For the recovery of a depleted stock, enforcement costs will be high at first but will decrease over time and can be recovered over a period of time as industry benefits increase [14,57]. Recovering enforcement costs from fishing industries not only increases the financial resources available for enforcement, but it reduces the burden of cost to non-benefitting community members and creates a positive feedback loop because fishers are more likely to demand efficiency in services they are funding [14,57]. Larger-scale fisheries in the United States, Australia, New Zealand, Iceland, and Canada have all successfully implemented mechanisms to recover some portion of enforcement costs from the fishing industry [15,31]. The fishing industry in Australia became more involved in the management process when fishing industry benefits were used to finance enforcement and management, which resulted in higher acceptance and compliance of management measures by fishermen [58]. In New Zealand, recovering enforcement costs from the fishing industry led to an increase in enforcement effort over time without an increase in costs because the fishing industry was incentivized to become more involved in the enforcement process [57]. It should be noted that these countries have robust institutions that allow cost recovery to be successful, which may not be the case for many small-scale fisheries of the developing tropics – this will be discussed further in the assumptions section below. However, these examples could still serve as bright spots and provide valuable lessons for designing and implementing such financing mechanisms in new places. A limited number of cost recovery programs do exist in smaller-scale scale fisheries, including those in Uganda and Namibia [32,33]. In the case of Caribbean spiny lobster fisheries, the Barbuda Council established a special coastal and fisheries management account where fines and fees associated with ocean resource use can be deposited in 2015 – an important step towards a sustainably financed enforcement system [personal communication].

The tradeoffs between financing mechanisms should be carefully considered in light of fishery characteristics and management goals [14,59]. The costs and feasibility associated with the mechanism, how the mechanism will distribute costs among users, and incentives that may be generated should be determined [59]. For example, a license or fishery participation fee may be relatively simple and cheap to collect, but when a wide range of fishing capacity exists between participants, smaller-scale fishers often suffer a disproportionate loss in profits and may be forced out of the fishery [32]. In other cases, a participation fee may have the desired effect of limiting participation in a fishery [14]. A mechanism to tax a fishers' input (effort) or output (catch) unit is more likely to evenly distribute the costs of enforcement among participants [14]. However, input taxes may unintentionally result in a total fishing effort increase because fishers are incentivized to increase untaxed effort units to make up for lost profits [32]. In this study, the landings tax was the largest available financing revenue stream for archetypes 1 and 3. A landings tax has been an effective method of recovering enforcement costs in fisheries with an adequate level of monitoring and enforcement [60]. This approach, however, assumes that landings are being fully and accurately reported, and may create incentives for underreporting, which occurred in Tanzania [61]. Export taxes are another option for cost recovery, but also have unintended consequences; such a tax may negatively impact the value of the fishery if it causes sellers to sell a larger portion of catch locally to avoid the export tax, thus flooding the market and reducing product value [32].

In this analysis, the break-even point was highly dependent on the archetype and types of stakeholders contributing to cost recovery, especially at low stock conditions. This demonstrates the importance of having all benefiting parties, not just fishing industries, contribute to enforcement cost recovery [60]. This is an especially important consideration for areas where the poverty level of fishers makes recovering enforcement costs through the fishing industry alone unfeasible [62]. These same areas often lack financial resources to adequately protect their marine resources [30] yet marine related-tourism can offer an opportunity to improve the local economy [59,63]. For example, in Mozambique a survey of divers revealed that the majority of dive tourists were attracted to the area because of the healthy marine ecosystem and the presence of certain species [64]. However, the species that attracted the divers were also targeted by a fishery with poorly enforced management. In this case, using revenue generated from the tourism industry to finance fisheries enforcement would likely yield large benefits to both industries.

It is not uncommon for managers to use revenue generated from the tourism industry to finance the costs of enforcement of a designated spatial area or marine reserve [59,65]. However, the surrounding fisheries may not receive spillover benefits from these protected areas if management and enforcement outside of designated spatial areas is inadequate. If revenue generated from tourism associated with a marine reserve could be used to finance fisheries enforcement outside of the marine reserve as well, the benefits of marine reserves to fisheries would likely increase. Although the literature does not show documented examples of how tourism revenue can be used to enforce fisheries regulations within fished areas, some studies have shown that tourists may be willing to pay more for diving [62,63] – this additional revenue could be used to pay for enforcement of management within fished areas.

4.3. Assumptions

Importantly, there are a number of assumptions that if changed could impact the results of the model. First, while we assume that fishers are purely profit-maximizing and that their expected benefits are influenced by the probability of being prosecuted and fined, there are other factors that might influence fishing behavior including moral principles and social pressure/norms [26,36]. Where appropriate, the social costs of illegal fishing could be examined by adding an additional cost parameter to the expected benefit function [24,26]. Other factors that might affect fisher behavior include perceived fairness, perceived legitimacy of rules and the management authority, and fisher involvement in management design and implementation [37]. Co-management and other arrangements such as cooperatives [38] that incorporate fisher participation in the design and implementation of management can strengthen perceived fairness, and possibly therefore compliance rates [37]. These considerations can especially relevant when designing new management and enforcement systems in small-scale fisheries. Moreover, the existence of these behavior-influencing factors suggests that there might be other ways to influence compliance rates besides traditional monitoring and enforcement, such as implementing programs that influence ethical codes and strengthen user participation, perceived fairness, and perceived legitimacy [26]. This could be a cost-effective alternative or supplement to increased enforcement in small-scale resource-limited fisheries.

A number of institutional assumptions are also made that might be violated, especially in small-scale fisheries with limited infrastructure. First, it is assumed that institutions are in place to facilitate the collection of license fees, taxes, and fines, and that a social planner is able to use these funds on enforcement costs. In reality, these management institutions might not exist, and are associated with implementation costs and operating costs of their own which were not included in this model. Second, it is assumed that the cost of prosecution and fine collection is zero, when in reality the prosecution costs can be substantial and may include attorney or court costs [66]. It is also assumed that illegal fishers who are caught are prosecuted, convicted, and fined, and that their fines are immediately collected. This is likely not the case in many fisheries in the developing tropics. One potential problem is that many fishers simply lack the financial resources to pay the fine. Another problem is low levels of prosecution and conviction rates [66]. To examine this in the model, one could add an additional probability parameter to the expected benefits function that would make expected cost of illegally fishing dependent on the probability of being prosecuted in addition to the probability of being detected and the level of the fine [13]. Since the likelihood of being prosecuted is likely less than one (which may be the case for example in some regions where the Corruption Perceptions Index is high), it is easy to imagine scenarios in which fishers are relatively undeterred by the threat of enforcement. In these cases, social benefits can decline, stocks can become or remain overfished, and industry profits can suffer. These potential issues suggest that broader institutional and political changes may be needed before an enforcement program is able to operate effectively.

5. Conclusion

This study presents a framework that can be used by managers to help design economically optimal enforcement systems, and pay for that enforcement, in small-scale fisheries settings where financial resources are often limited. By using an illustrative Caribbean spiny lobster fishery representative of the situation in Barbuda, the results demonstrate that optimal enforcement effort should be highest for low biomass, and can decrease as the stock rebuilds. Optimal enforcement also depends on the ecosystem services and stakeholders receiving benefits from those services, and is higher when dive tourism (or other sector that values biomass of fish in the water) is present. In depleted or collapsed initial stock conditions, the social planner revenue will not be sufficient to contemporaneously finance the optimal enforcement effort, which suggests that additional capital investment will likely be needed at the onset of new enforcement strategies. However, the social planner can eventually pay back external investors in full, as well as sustainably finance ongoing enforcement efforts through the use of a landings tax, enforcement fines, fisher licensing fee, and a dive tourism tax.

In 2015, the island of Barbuda took a final, important step for implementing the fishery and zoning regulations established in August 2014 by developing and implementing an enforcement strategy. The island obtained a new fisheries patrol vessel and law enforcement agents from four agencies gathered in Barbuda to train and collaborate on the enforcement of ocean laws in the island's waters in August 2014. Although the implementation of a successful cost-recovery program to finance fisheries enforcement is not certain, the Barbuda Council has been given the authority to establish a special account for Barbuda coastal and fisheries management where fines and fees associated with ocean resource use can be deposited. As more data and information becomes available in Barbuda and other small-scale fisheries, the framework presented in this study may serve as a useful tool to help fishery managers develop and finance an optimal enforcement strategy.

Acknowledgment

We would like to acknowledge the generous financial support from the Waitt Institute for conducting this research.

References

[1] D.W. Evans, The consequences of illegal, unreported and unregulated fishing for fishery data and management, FAO Fisheries Rep. 12 (2001) 222–233.

[2] U.R. Sumaila, J. Alder, H. Keith, Global scope and economics of illegal fishing, Mar. Policy 30 (2006) 696–703, http://dx.doi.org/10.1016/j.marpol.2005.11.001.

[3] D.A. Varkey, C.H. Ainsworth, T.J. Pitcher, Y. Goram, R. Sumaila, Illegal, unreported and unregulated fisheries catch in Raja Ampat Regency, Eastern Indonesia, Mar. Policy 34 (2010) 228–236, http://dx.doi.org/10.1016/j.marpol.2009.06.009.

[4] D.J. Agnew, J. Pearce, G. Pramod, T. Peatman, R. Watson, J.R. Beddington, T. J. Pitcher, Estimating the worldwide extent of illegal fishing, PLoS One 4 (2009) e4570, http://dx.doi.org/10.1371/journal.pone.0004570.

[5] D. Pauly, D. Zeller, Catch reconstructions reveal that global marine fisheries catches are higher than reported and declining, Nat. Commun. (2016), http://dx.doi.org/10.1038/ncomms10244.

[6] C. Safina, D.H. Klinger, Collapse of bluefin tuna in the Western Atlantic, Conserv. Biol. 22 (2008) 243–246, http://dx.doi.org/10.1111/j.1523-1739.2008.00901.x.

[7] I.C. Field, M.G. Meekan, R.C. Buckworth, C.J. Bradshaw, Protein mining the world's oceans: Australasia as an example of illegal expansion-and-displacement fishing, Fish Fish. 10 (2009) 323–328, http://dx.doi.org/10.1111/j.1467-2979.2009.00325.x.

[8] B. Öztürk, Some remarks of Illegal , Unreported and Unregulated (IUU) fishing in Turkish part of the Black Sea, J. Black Sea/Medit. Environ. 19 (2013) 256–267.

[9] C.C. Schmidt, Economic drivers of Illegal, Unreported and Unregulated (IUU) fishing, Int'l. J. Mar. Coast. L 20 (2005) 479–507, http://dx.doi.org/10.1163/157180805775098630.

[10] B. Le Gallic, A. Cox, An economic analysis of Illegal, Unreported and Unregulated (IUU) fishing: key drivers and possible solutions, Mar. Policy 30 (2006) 689–695, http://dx.doi.org/10.1016/j.marpol.2005.09.008.

[11] FAO, The State of World Fisheries and Aquaculture: Opportunities and Challenges, Food and Agriculture Organization of the United Nations, Rome, 2014.

[12] FAO, report of the expert workshop to estimate the magnitude of illegal, unreported and unregulated fishing globally, FAO Fisheries and Aquaculture Report 1106, Rome 2015.

[13] R. Arnason, Fisheries management and operations research, Eur. J. Oper. Res. 193 (2009) 741–751, http://dx.doi.org/10.1016/j.ejor.2007.07.028.

[14] P. Wallis, O. Flaaten, Fisheries Management Costs: Concepts and Studies, in: proceedings of the 10th IIFET Conference. Corvallis, 2001.

[15] OECD, The Costs of Managing Fisheries, Org. Econ. Co-op. Develop. Publish., Paris (2003).

[16] J.R. Beddington, R.B. Rettig, Approaches to the regulation of fishing effort, FAO Fish. Tech. Paper 243 (1984) 1–39.

[17] G.A. Petrossian, Preventing Illegal, Unreported And Unregulated (IUU) fishing: a situational approach, Biol. Conserv. 18 (2014) 39–48, http://dx.doi.org/10.1016/j.biocon.2014.09.005.

[18] K. Davis, M. Kragt, S. Gelcich, S. Schilizzi, D. Pannell, Accounting for enforcement costs in the spatial allocation of marine zones, Conserv. Biol. 29 (2015) 226–237, http://dx.doi.org/10.1111/cobi.12358.

[19] P. Guidetti, et al., Italian marine reserve effectiveness: does enforcement matter? Biol. Conserv. 141 (2008) 699–709, http://dx.doi.org/10.1016/j.biocon.2007.12.013.

[20] L.M. Gigliotti, W.W. Taylor, The effect of illegal harvest on recreational fisheries, N. Am. J. Fish. Manag. 10 (1990) 106–110, http://dx.doi.org/10.1577/1548-8675(1990)010 < 0106:teoiho > 2.3.co;2.

[21] A. Balmford, et al., Economic reasons for conserving wild nature, Science 297 (2002) 950–953, http://dx.doi.org/10.1126/science.1073947.

[22] C. Pet-Soede, H.S.J. Cesar, J.S. Pet, An economic analysis of blast fishing on Indonesian coral reefs, Environ. Conserv. 27 (1999) 83–93, http://dx.doi.org/10.1017/S0376892999000132.

[23] F. Le Manach, C. Gough, A. Harris, F. Humber, S. Harper, D. Zeller, Unreported fishing, hungry people and political turmoil: the recipe for a food security crisis in Madagascar? Mar. Policy 36 (2012) 218–225, http://dx.doi.org/10.1016/j.marpol.2011.05.007.

[24] K.K. Viswanathan, N.M. Abdullah, I. Susilowati, I.M. Siason, C. Ticao, Enforcement and compliance with fisheries regulations in Malaysia, Indonesia and the Philippines. Proc. Int'l Workshop Fisher. Co-manag. 5, 1997, pp.1–25, uri: ⟨http://hdl.handle.net/10535/7871⟩.

[25] R. Hilborn, J.M. Orensanz, A.M. Parma, Institutions, incentives, and the future of fisheries, Philos. Trans. R. Soc. B 360 (2005) 47–57, http://dx.doi.org/10.1098/rstb.2004.1569.

[26] A.K. Kuperan, J.G. Sutinen, Blue water crime: deterrence, legitimacy, and compliance in fisheries, Law Soc. Rev. 32 (1998) 309–338, http://dx.doi.org/10.2307/827765.

[27] C.H. Ainsworth, H.N. Morzaria-Luna, I.C. Kaplan, P.S. Levin, E. Fulton, Full compliance with harvest regulations yields ecological benefits: Northern Gulf of California case study, J. Appl. Ecol. 49 (2012) 63–72, http://dx.doi.org/10.1111/j.1365-2664.2011.02064.x.

[28] R. Tinch, L. Dickie, B. Lanz, Costs of Illegal, Unreported and Unregulated (IUU) Fishing in EU Fisheries, Economics for the Environment Consultancy, London, 2008.

[29] I. Boto, C. La Peccerella, S. Scalco, Fighting Against Illegal, Unreported and Unregulated Fishing (IUU): Impacts and Challenges for ACP Countries, 16th Brussels, Development Briefing on Illegal, Unreported and Unregulated Fishing, Brussels, 2009.

[30] K.L. Cochrane, N.L. Andrew, A.M. Parma, Primary fisheries management: a minimum requirement for provision of sustainable human benefits in small-scale fisheries, Fish Fish. 12 (2011) 275–288, http://dx.doi.org/10.1111/j.1467-2979.2010.00392.x.

[31] A.A. Brinson, E.M. Thunberg, 2013. The Economic Performance of U.S. Catch Share Programs, US Dep. Commer, NOAA Tech. Memo, NOAA-TM NMFS-F/SPO-133a.

[32] B.B. Keizire, 2001. Opportunities and options for financing fisheries management in Uganda, UNU-Fisheries Training Programme Fellows Document, Reykjavik.

[33] P. Manning, The namibian hake fishery, in: S. Cunningham, T. Bostock (Eds.), Successful Fisheries Management: Issues, Case Studies and Perspectives, Eburon Academic Publishers, The Netherlands, 2005.

[34] R. Arnason, 2006. Fisheries enforcement: basic theory, Proc. 13th IIFET Conf., Portsmouth.

[35] S.S. Gezelius, Do norms count? state regulation and compliance in a Norwegian fishing community, Acta Sociol. 45 (2002) 305–314, http://dx.doi.org/10.1177/000169930204500404.

[36] S.S. Gezelius, M. Hauck, Toward a theory of compliance in state-regulated livelihoods: a comparative study of compliance motivations in developed and developing world fisheries, Law Soc. Rev. 45 (2011) 435–470, http://dx.doi.org/10.1111/j.1540-5893.2011.00436.x.

[37] M. Hauck, Small-scale fisheries compliance: integrating social justice, legitimacy and deterrence, in: R.S. Pomeroy, N. Andrew (Eds.), Small-scale Fisheries Management: Frameworks and Approaches for the Developing World, CAB International, Cambridge, http://dx.doi.org/10.1079/9781845936075.0196.

[38] D.A. Ovando, et al., Conservation incentives and collective choices in cooperative fisheries, Mar. Policy 37 (2013) 132–140, http://dx.doi.org/10.1016/j.marpol.2012.03.012.

[39] R. Arnason, On optimal dynamic fisheries enforcement, Mar. Resour. Econ. 28 (2013) 361–377, http://dx.doi.org/10.5950/0738-1360-28.4.361.

[40] I.S. Horsford, H. Simon, M. Archibald, J. Webber, T. Joseph, Biology, status and current management of the Caribbean spiny lobster (*Panulirus argus*) in Antigua and Barbuda, Proc. Gulf Caribb. Fish. Inst. 66 (2013) 423–433.

[41] M. Bigue, O. Rosero, 2015. Barbuda Blue Halo Enforcement Blueprint, WildAid. ⟨http://barbuda.waittinstitute.org/wp-content/uploads/2014/09/Barbuda-Blue-Halo-Enforcement-Blueprint-WildAid.pdf⟩.

[42] M. Haddon, Modeling and Quantitative Methods in Fisheries, Second Edition, CRC Press, Boca Raton, 2010.

[43] COBECOS, 2009. Costs and Benefits of Control Strategies, EU Sixth Framework Programme EU-FP-044153, Brussels.

[44] E. Sala, C. Costello, D. Dougherty, G. Heal, K. Kelleher, J.H. Murray, A. A. Rosenberg, R. Sumalia, A general business model for marine reserves, PLoS One 8 (2013) e58799, http://dx.doi.org/10.1371/journal.pone.0058799.

[45] R.A. Morris, A bioeconomic analysis of the Jamaican industrial spiny lobster (*Panulirus argus*) fishery, United Nations University: Fisheries Training Programme, Reykjavik, 2010.

[46] I.S. Horsford, Economic viability of marine capture fisheries in Antigua and Barbuda- A case study, FAO Fisheries Report 640 (2002) 101–117.

[47] E. Green, R. Donnolley, Recreational scuba diving in Caribbean marine protected areas: do the users pay? AMBIO 32 (2003) 140–144, http://dx.doi.org/10.1579/0044-7447-32.2.140.

[48] C.M. Holmlund, M. Hammer, Ecosystem services generated by fish populations, Ecol. Econ. 29 (1999) 253–268, http://dx.doi.org/10.1016/S0921-8009(99)00015-4.

[49] E.G. Oracion, M.L. Miller, P. Christie, Marine protected areas for whom? fisheries, tourism, and solidarity in a Philippine community, Ocean Coast. Manag. 48 (2005) 393–410, http://dx.doi.org/10.1016/j.ocecoaman.2005.04.013.

[50] H.S.J. Cesar, P.J.H. van Beukering, Economic valuation of the coral reefs of Hawai'i, Pac. Sci. 58 (2004) 231–242, http://dx.doi.org/10.1353/psc.2004.0014.

[51] S.F. Farrow, Marine protected areas: emerging economics, Mar. Policy 20 (1996) 439–446, http://dx.doi.org/10.1016/j.ocecoaman.2005.04.013.

[52] R.F. Subade, Mechanisms to capture economic values of marine biodiversity: The case of Tubbataha Reefs UNESCO World Heritage Site, Philippines, Mar. Policy 31 (2007) 135–142, http://dx.doi.org/10.1016/j.marpol.2006.05.012.

[53] L. Ledoux, R.K. Turner, Valuing ocean and coastal resources: a review of practical examples and issues for further action, Ocean Coast. Manag. 45 (2002) 583–616, http://dx.doi.org/10.1016/S0964-5691(02)00088-1.

[54] J.P.G. Spurgeon, The economic valuation of coral reefs, Mar. Pollut. Bull. 24 (1992) 529–536, http://dx.doi.org/10.1016/0025-326X(92)90704-A.

[55] G. Sutinen, P. Andersen, The economics of fisheries law enforcement, Land Econ. 61 (1985) 387–397, http://dx.doi.org/10.2307/3146156.

[56] R. Arnason, Economic instruments for achieving ecosystem objectives in fisheries management, ICES J. Mar. Sci. 57 (2000) 742–751, http://dx.doi.org/10.1006/jmsc.2000.0733.

[57] M. Harte, Funding commercial fisheries management: lessons from New Zealand, Mar. Policy 31 (2007) 379–389, http://dx.doi.org/10.1016/j.marpol.2006.11.002.

[58] A. Cox, 2001. Cost recovery in fisheries management: the Australian experience, Proc. 10th IIFET Conf., Corvallis.

[59] B.D. Blackwell, D.R. Brumbaugh, C.P. Dahlgren, 2013. Sustainable Finance Analysis for the South Berry Islands Marine Reserve, Report submitted to the Nature Conservancy Northern Caribbean Program, Nassau.

[60] R. Metzner, 2007. Report of the expert consultation on low-cost fisheries management strategies and cost recovery, FAO Fisheries and Aquaculture Report 853.

[61] C. Boyd, A. Inamdar, Sustainable Financing of Coastal Management Activities in Eastern Africa, SEACAM, Oxford, 2001.

[62] K. Reid-Grant, M.G. Bhat, Financing marine protected areas in Jamaica: an exploratory study, Mar. Policy 33 (2009) 128–136, http://dx.doi.org/10.1016/j.marpol.2008.05.004.

[63] S.M. Thur, User fees as sustainable financing mechanisms for marine protected areas: an application to the Bonaire National Marine Park, Mar. Policy 34 (2010) 63–69, http://dx.doi.org/10.1016/j.marpol.2009.04.008.

[64] Y. Tibirica, Y., A. Birtles, P. Valentine, D.K. Miller, Diving tourism in Mozambique- an opportunity at risk? Tour. Mar. Environ. 7 (2011) 141–151, http://dx.doi.org/10.3727/154427311x13195453162732.

[65] J. Gibson, M. McField, S. Wells, Coral reef management in Belize: an approach through integrated coastal zone management, Ocean Coast. Manag. 39 (1998) 229–244, http://dx.doi.org/10.1016/S0964-5691(98)00007-6.

[66] W.J. Furlong, The deterrent effect of regulatory enforcement in the fishery, Land Econ. 67 (1991) 116–129, http://dx.doi.org/10.2307/3146490.