



Destructive fishing and fisheries enforcement in eastern Indonesia

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ABSTRACT: A simple bioeconomic leader–follower model was constructed to simulate snapper (family Lutjanidae) and grouper (family Serranidae) fisheries in Raja Ampat, Indonesia, an area of significant coral and fish biodiversity. We developed a leader–follower game, wherein the Regency government as the leader chooses an enforcement model to discourage illegal fishing. Fishers are then given a choice to fish using legal gears, such as handlines, or to fish with illegal gears, e.g. dynamite (for snapper) or cyanide (for grouper). Given prices and costs of legal and illegal fishing, the status quo simulations with no Regency enforcement result in a large amount of illegal catch throughout the 50 yr simulation, which agrees with expert opinion that destructive illegal fishing is occurring in the region. In an attempt to include ecosystem-based management principles into Raja Ampat governance, we introduce an enforcement regime in the form of detecting and punishing illegal fishing. Results suggest that current fishing practices do not account for the disproportionate ecosystem effects of destructive fishing, and that elimination of dynamite fishing may be easier for the government due to the high profitability of the live fish trade connected with cyanide fishing.

KEY WORDS: Coral reef conservation · Ecosystem-based management · Leader–follower · Incentives · Live reef trade

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INTRODUCTION

It is clear that traditional methods of marine resource management have focused only on the short-term benefits of commodity production (Berman & Sumaila 2006), and therefore have not resulted in sustainable fisheries. Consequently, there is a need for alternative management schemes which incorporate socio-economic, political, and ecological factors into decision making (Gislason et al. 2000). Ecosystem-based management (EBM) is an integrated science-based approach to management that takes multiple scales and time periods into account (Leslie & McLeod 2007). EBM is also an adaptive approach that strives to balance societal objectives, and makes an explicit link between people and the ecosystem (Sumaila 2005). As such, EBM has the potential to

address the varying and complex dynamics of resource management (Pikitch et al. 2004). The Raja Ampat Regency in Papua Province, Indonesia, is currently developing and implementing a sustainable fisheries management system, which includes a desire to incorporate EBM principles and objectives into decision making (Huffard et al. 2012).

Raja Ampat boasts the world's highest coral reef biodiversity, with 75% of known hard coral species found in the area (Halim & Mous 2006) and is home to over 1200 species of fish (Ainsworth et al. 2008). Artisanal fishing in this region, and in Indonesia in general, is an important economic sector (Dohar & Anggraeni 2007), but the introduction of new gears can throw off the traditional balance that artisanal fisheries often encompass (Kusuma-Atmadja & Purwaka 1996). Although eastern Indonesia boasts

reefs with more live coral than other parts of Indonesia, the use of destructive illegal fishing gears, mainly explosives and cyanide, is among the major threats to sustainable fisheries in the Raja Ampat Regency (Halim & Mous 2006) and to the implementation of EBM more broadly.

Illegal, unreported, and unregulated (IUU) fishing is gaining attention around the world, with fisheries scientists listing it as a major barrier to the sustainability of marine resource use (Pitcher et al. 2002, Sumaila et al. 2006), and this is certainly true of destructive fishing in Indonesia (Pet-Soede & Erdmann 1998, Pet-Soede et al. 1999, Halim & Mous 2006). IUU fishing can undermine management programs (FAO 2002) because it can lead to underestimation of catch and effort (Pitcher et al. 2002). Furthermore, dynamite and cyanide fishing can negatively affect fish habitat, and are thus inherently unsustainable fishing methods (Pauly 1989).

In an era of EBM, the ability to model multiple users of an ecosystem is becoming increasingly important (Sumaila 2005). Differences in resource user uncertainty, rates of discount, and risk aversion can impede sustainable fisheries (Sumaila 2005), a core EBM goal. To help elucidate the factors that impede or assist in the collaborative management process, and to help policy makers understand why EBM is such a hard goal to reach, economists explicitly model the expected utility functions of user groups. Leader–follower games model situations where 1 player in the game (the leader) makes the move first by setting out the management regime and the rules under which fishing takes place. In the following stages, the other players (the followers) choose their course of action. We developed a leader–follower Stackelberg model to compare the profitability of illegal (destructive) and legal fishing methods in Raja Ampat. Specifically, we analyzed how efforts taken by the government in the form of the probability of detecting illegal fishers, and the penalty faced by violators (see Becker 1968), could act as disincentives to illegal fishing in Raja Ampat. In doing so, we also drew on the relevant fisheries enforcement literature.

The following section provides a brief background on fisheries in Raja Ampat. We also include an overview of the literature on fisheries enforcement and the relevance of leader–follower games. This is followed by the development of a bioeconomic model to simulate the behavior of legal and illegal fishers in Raja Ampat, in the absence and presence of an enforcement program. We then discuss the results of the analysis and offer some concluding remarks.

BACKGROUND

Raja Ampat's artisanal fisheries

The artisanal fisheries sector in Raja Ampat was valued at Rp63 billion Indonesian Rupiah in 2006, equivalent to about US \$7 million (Dohar & Anggraeni 2007). It is a mixed-species fishery, with several target species pursued and fishers employing several gear types. For example, a fisher may fish at any time with a handline or spear and target snapper (Lutjanidae), grouper (Serranidae), or trevally (Siganidae), among other fish. Legal fishing gears used for artisanal fishing include handline, dip net, gill net, permanent trap, and spear/harpoon. The average artisanal fisher in Raja Ampat fishes about 15 d mo⁻¹ (Dohar & Anggraeni 2007) and earns about US \$1000 yr⁻¹ (Bailey et al. 2008).

Fishing with the use of explosives and cyanide also occurs in Raja Ampat. Pet-Soede & Erdmann (1998) reported that the low population densities in eastern Indonesia make monitoring and enforcement difficult. Blast and cyanide fishing are used to catch reef-associated fish, with snapper (dynamite), grouper, and Napoleon wrasse *Cheilinus undulatus* (cyanide) being the main targets (Pet-Soede et al. 1999). When Halim & Mous (2006) asked households in Raja Ampat whether family members engaged in destructive fishing practices, all respondents said 'no'. However, most fishers with whom we spoke during our field trips in Raja Ampat admitted that they heard blast fishing every day. Reefs exposed to repeated blasts 'are often reduced to little more than shifting rubble fields' (Pet-Soede & Erdmann 1998, p. 29). Blast fishing occurs with homemade fertilizer bombs (Pet-Soede et al. 1999), which means the cost of making the bombs is probably much lower than it once was, when fishers used actual dynamite. Blast fishers in large operations can make between US \$50 and 150 wk⁻¹ (Pet-Soede & Erdmann 1998), while the small-scale blast fishers make about US \$14 wk⁻¹ (Pet-Soede et al. 1999).

The current management regime states that fishers must be caught in the act of cyanide fishing in order to be charged with illegal fishing (M. Erdmann pers. comm.). The result is that regulators are powerless if they find a fisher with cyanide and live fish in his boat (note that although women do participate in some fishing activities, such as coastal gleaning, boats are generally owned and operated by men). The discussion regarding how much reef damage cyanide fishing causes varies widely, but quantitative simulations suggest that the worst-case scenario could

result in a loss of 9.5% coral cover per year (Saila et al. 1993). Furthermore, the high catch per unit effort of cyanide fishing on grouper spawning aggregations can quickly lead to overfished populations (Mous et al. 2000). The price for live fish caught using cyanide varies, but some fish, such as the coral trout (the grouper species *Plectropomus leopardus*), can fetch up to US \$19 kg⁻¹ (Pet-Soede & Erdmann 1998).

Fisheries enforcement

Some amount of regulation in fisheries, in the form of monitoring, control, and surveillance (MCS), is required to counteract an array of externalities and free-rider incentives (Sutinen & Kuperan 1999). In Indonesia, effective regulation is hampered by a lack of funding, dispersed fishing and landing sites, and a decentralized system that shares regulatory oversight between different hierarchies within the government. While these limitations are real barriers to fisheries regulation, the Indonesian government recognizes the value that artisanal and commercial fisheries bring to communities and the economy, and thus has prioritized improvements in MCS across the country, specifically MCS aimed at deterring IUU fishing (FAO 2007).

Regulation is costly, and the benefits of such programs can be marginal (Sutinen & Andersen 1985); however, research also demonstrates that decreases in enforcement investment lead to suboptimal conservation returns (Keane et al. 2008). Together, these 2 statements lead to the conclusion that enforcement is necessary for conservation, but its costs can be prohibitive. This is true for all countries, but for developing world countries, such as Indonesia, this may be especially true given real limitations on government financial capital dedicated to fisheries MCS (Balmford & Whitten 2003). This has led to increasing participation by the private sector, particularly by non-profit organizations, as well as by public international institutions, in contributing to conservation activities throughout Indonesia (Bailey et al. in press).

Regulatory initiatives can be constructed in several ways. In this paper, we model regulation through the probability of detecting illegal fishing (monitoring and compliance) and the implementation of a penalty fee (enforcement). This is similar to the approach taken by Sutinen & Andersen (1985) in their seminal fisheries enforcement analysis, and subsequently by Furlong (1991), who empirically tested non-compliance and perceptions on enforcement. Practically speak-

ing, the costs of imposing a penalty (enforcement) are generally thought to be less than the costs required to detect illegal activity in the first place (monitoring costs) (Sutinen & Andersen 1985, Balmford & Whitten 2003), meaning that theoretically, governments would be better off maintaining a low-cost compliance system but extracting benefits through a high-value penalty fee. However, there are several reasons to avoid unnecessarily high penalty fees, with examples particularly relevant to Indonesia being when sanctions are socially costly, when corruption is likely, or when individual wealth varies greatly (see Balmford & Whitten 2003 for a review).

In this paper, we did not determine the optimal compliance and enforcement program from a cost effectiveness (or economic efficiency) standpoint. Rather, we determined and present a variety of compliance and enforcement combinations that could lead to complete elimination of illegal fishing. We believe it is vastly simplistic to assume that the optimal enforcement program for the government of Indonesia is one that occurs when the marginal cost to the government of enforcement is equal to the marginal benefit of the fishery. Rather, there are reasons to encourage coral reef conservation above and beyond simply breaking even, and we prefer to rely on a program that seeks economic improvement (and not optimization) as measured by fisheries performance. That being said, benefits from coral reef conservation accrue not only to the fisheries sector; so, determining the acceptable level of enforcement costs should not be derived solely from fisheries. We assume that in an EBM sense, the complete elimination of illegal fishing is optimal for society.

Leader–follower games

A leader–follower dynamic is modeled in this paper, which essentially represents a 2-stage game. This follows the approach taken by Charles et al. (1999), although those authors did not explicitly describe their model as a leader–follower game, and the work of Kronbak & Lindroos (2006). In the first analysis, the authors compared an unregulated fishery with a fishery regulated by input and output controls, as initiated by the leader, with regulation including a probability of detecting illegal fishing and a penalty function applied to illegal fishers (the followers; Charles et al. 1999), similar to that described by Sutinen & Andersen (1985). Avoidance behavior was also included in their model, and the authors concluded that illegal fishing can be re-

duced toward 0 with increasing levels of enforcement (Charles et al. 1999).

Kronbak & Lindroos (2006) set up a 4-stage model which involves states, as leaders in the game, determining coalition structures and an enforcement structure in the first 2 stages. In the final 2 stages, fishers choose their coalition structure and then optimize their effort to maximize profit. Their analysis has interesting implications for the institutional structures of governance (centralized versus decentralized) on compliance behavior of fishers (Kronbak & Lindroos 2006). Specifically with regard to the costs of enforcement, they found that cooperation among those setting the rules leads to higher compliance among fishers. In our model, we did not assume that several enforcement bodies must come together to cooperate, or that fishers can choose any form of coalition or cooperative structure among themselves. We relied on the leader–follower structure where an enforcement regime is implemented such that incentives exist for fishers to cooperate with regulations.

We assumed that the leader in the game is the Regency government, the formal legal ‘owner’ of marine resources in Raja Ampat. However, villagers tend to respect the authority of the traditional village over the formal government (Halim & Mous 2006). Based on his journey through the remote islands of eastern Indonesia, Severin (1997, p. 67) stated that ‘... the authority of these traditional leaders was more respected than the regulations which ultimately come from Jakarta ... exploitation of the land and sea should be done according to custom.’ Customary marine tenure rights are still enforced and respected in Raja Ampat. The traditional kinship groups are descendants from the first families in Raja Ampat, and these groups, present in each village, are the informal ‘owners’ of land and marine resources (A. Suebu pers. comm.).

The model developed herein assumes that the followers in the game are kinship groups, hereafter referred to as a village, who have the ability to control fisher actions due to their informal legitimacy. That is, the villages have 2 different fishing strategies available to them: legal or illegal, which they decide upon given the enforcement program initiated by the Regency.

Two leader–follower games were simulated here to evaluate the effort and profitability of illegal fishing, and the possible incentives that can be applied by the Regency. The first model considers blast fishing targeting snapper species. Using grouper species, the second model analyzes the cyanide fishery, and the

implications for effort, profit, and management. Both simulations are based on the same model developed below. The software package Powersim was used as a computational aide (Powersim Software).

THE MODEL

Population dynamics

A simple logistic growth model is used here to describe the biology of the system. This model assumes that, in an unfished population, change in the population biomass with time (t) is related to the intrinsic rate of growth of the stock, r , the stock’s carrying capacity, K , and the current stock size, B_t , as per the following equation:

$$\frac{dB_t}{dt} = rB_t \left(1 - \frac{B_t}{K}\right), \quad B_t \geq 0 \quad (1)$$

Eq. (1) implies that the growth in biomass is >0 so long as the current biomass is >0 and $<K$. This is a standard relationship in ecology, which, while simple, is adequate for the present model. Both cyanide and blast fishers are not particularly discriminatory in size selection of the catch, and therefore we argue that a simple logistic growth model, and not an age-structured model, is adequate. Further to this, we are interested in how the population grows from time 1 to time 2, so we are interested in the start and end weights, rather than how an individual or age cohort grows through its lifetime (see Gamito 1998 for a discussion on growth models).

A simple production model of the Cobb-Douglas form (Cobb & Douglas 1928) is used to simulate catch, h_t , where q is the catchability coefficient, which in this model is assumed constant over time (see Mackinson et al. 1997, for a general treatment of q). The catchability coefficient represents the proportion of the total biomass that is removed by 1 unit of effort in a given period. In this model, effort, E , is measured in number of trips per year, and must be ≥ 0 .

$$h_t = qE_t^\alpha B_t^\beta, \quad E_t \geq 0, B_t \geq 0 \quad (2)$$

In this equation, it is assumed that $\alpha = \beta = 1$, that is, there are constant returns to catch based on unit increases in effort or biomass. Hyperstability is implied if $\alpha < 1$, $\beta < 1$ (increases in biomass and/or effort result in less than equal catch increases), while $\alpha > 1$, $\beta > 1$ implies hyperdepletion (increases in biomass and/or effort result in greater than equal increases in catch) (Walters & Martell 2004). This simplified equation is known as the Schaefer catch equation (Schae-

fer 1954). This assumption is likely adequate for snapper, but for grouper, a family of fishes that does tend to aggregate, it could be that hyperdepletion is more likely. In this case, our results would be overly optimistic about the long-term benefits possible from continued illegal fishing, meaning that illegal fishing may be more detrimental than we allow for in the current study.

By incorporating the catch equation, Eq. (2), into Eq. (1), we get the following population dynamics of the stock:

$$\frac{dB_t}{dt} = rB_t \left(1 - \frac{B_t}{K}\right) - qE_t B_t, \quad E_t \geq 0, B_t \geq 0 \quad (3)$$

Strategic variables

Recall that the village is assumed to be able to allocate fisher effort to 1 of 2 types of fishing strategies: using legal or illegal gears. Let the type of strategy, s , be the set of these 2 types of fishing: $s = \{s, -s\}$, where s represents legal fishing and $-s$ represents illegal fishing.

Revenue

Total revenue, TR , is the product of the catch, h_t , and the unit price, P . Unit price is assumed to be constant over time; however, catches with different fishing strategies can command different prices. We can describe the single-period total revenue for a given strategy as:

$$TR_{s,t} = P_s q E_{s,t} B_t, \quad \forall s, t \quad (4)$$

with the total revenue of that strategy through time being:

$$TR_s = \sum_{t=0}^T TR_{s,t}, \quad \forall s \quad (5)$$

and the total revenue to the village over time and over both strategies as:

$$TR = TR_s + TR_{-s} \quad (6)$$

Cost

We assume perfectly malleable capital in this model (Sumaila 1997), i.e. the capital investment for the boat is a sunk cost (the fisher has already paid for the vessel, whether he fishes or not), and the same vessel

is used for either type of fishing strategy. Fishing effort can therefore be easily allocated to either strategy on a trip by trip basis. Therefore, only variable costs are considered in this model.

The total cost, TC , of fishing is the product of the effort, E , and the unit variable cost of effort, co , and is modeled as an 'almost' linear function (Cowell 1986, Madden 1986, Sumaila 1995). The unit cost of effort is assumed constant through time. Let the single-period cost of a given catch strategy be:

$$TC_{s,t} = \frac{co_{s,t} E_{s,t}^{1+b}}{1+b}, \quad \forall s, t \quad (7)$$

here, as b approaches 0, the cost function is almost linear. This introduces concavity in the profit function, thus ensuring convergence to a solution (Sumaila 1995).

The total cost of fishing using a given strategy over time is:

$$TC_s = \sum_{t=0}^T TC_{s,t}, \quad \forall s \quad (8)$$

and the total cost of fishing over time and over both strategies is computed as:

$$TC = TC_s + TC_{-s} \quad (9)$$

One more cost must be factored in, namely, the expected cost when caught engaging in illegal fishing (i.e. the penalty, Pen). This cost is assumed to be a function of the monitoring and enforcement plan put in place by the Regency government:

$$Pen_t = \rho \times Fee \times E_{-s,t} \quad (10)$$

In effect, the expected cost is the product of the probability of being apprehended, ρ , the penalty imposed when apprehended, Fee , and the amount of illegal effort. Note that any avoidance costs borne by the fishers are not considered in this analysis, and thus are assumed to be 0.

Therefore, the single period cost of fishing illegally is:

$$TC_{-s,t} = \frac{co_{s,t} E_{s,t}^{1+b}}{1+b} + \rho \times fee \times E_{-s,t}, \quad \forall t \quad (11)$$

with the total cost over time expressed as:

$$TC_{-s} = \sum_{t=0}^T TC_{-s,t} \quad (12)$$

Net benefit

The single period (private) net benefit to the village, π , is therefore the sum of the difference

between the total revenue and the total cost of each fishing strategy in a given period:

$$\pi_t = (TR_{s,t} - TC_{s,t}) + (TR_{-s,t} - TC_{-s,t}) \quad (13)$$

with the discounted total net benefit over time calculated as:

$$\pi = \sum_{t=0}^T \delta^t (TR_{s,t} - TC_{s,t}) + \sum_{t=0}^T \delta^t (TR_{-s,t} - TC_{-s,t}) \quad (14)$$

where δ is the discount factor, and is equal to $(1 + dr)^{-1}$, where r is the discount rate of the village (in this model we have assumed a discount rate of 7%, but this is explored in the sensitivity analysis). By expanding the revenue and cost functions, we see that the total net benefit to the village, over time and considering both catch strategies, is:

$$\pi = \sum_{t=0}^T \delta^t \left(qE_{s,t} B_t P_s - \frac{cO_s E_{s,t}^{1+b}}{1+b} \right) + \sum_{t=0}^T \delta^t \left(qE_{-s,t} B_t P_{-s} - \frac{cO_{-s} E_{-s,t}^{1+b}}{1+b} \right) - \rho \times Fee \times E_{-s,t} \quad (15)$$

This benefit is in fact a private one, where the costs and benefits only of the fishery are considered in decision making. However, by way of the α parameter, we do allow for ecosystem signals to be seen (see the solution algorithm in the Appendix to see how alpha is incorporated into the simulations).

Strategic interaction in our model takes place in 2 ways: firstly, between fishers where there is no enforcement, and here the interaction is due to stock effects; and secondly, strategic interaction takes place between leaders and followers, where the optimal choice of the government as leader is to implement an enforcement regime that creates the necessary incentives for fishers to behave in a certain way. In this way, the action of the leader has implications for the actions of the followers (and vice versa, in setting the enforcement regime in the first place).

Externalities and EBM

Externalities in the system are introduced with the term α , which describes the relative impact of illegal fishing on the reef system. Destructive fishing practices, such as blast and cyanide fishing, alter the marine habitat, which can negatively affect both target and non-target species. The effects of destructive fishing could impact reefs to such a state that recovery from cyanide and explosives does not occur for over 2 decades (Saila et al. 1993, Fox & Caldwell 2006). As such, the model was modified to incorpo-

rate this disproportionate impact on the stock. This impact has consequences for fishers themselves, but is also a way of incorporating the societal costs of illegal fishing. Not only do fishers lose out in the long term from destructive fishing practices, but deleterious impacts on the reef itself will likely lead to societal costs above and beyond decreased fisheries production.

We investigated the impact that varying the EBM externalities has via the α parameter. In the case that fishers only consider direct fishing impacts and no ecosystem impacts, $\alpha = 1$, then the impact of legal and illegal fishing is equal. However, the impacts of destructive fishing are an externality that villages must in fact incorporate in their present-day fishing decisions and that society as a whole considers. Refer to the Appendix to see how α is incorporated into the solution algorithm. In an effort to use the precautionary approach and in keeping with EBM (currently the goal of the Raja Ampat Regency government), we chose to use $\alpha = 2$ for both the baseline and our 'optimal' solutions, and this assumption is discussed in the sensitivity analysis.

Optimization

We assume that the objective of the village is to decide on a sequence of effort through time, using legal and/or illegal methods, to maximize their net benefit, π , or discounted economic rent, through time, subject to the obvious constraints. This model represents a 2-stage leader-follower game. In step 1, the Regency government (leader) chooses its monitoring and enforcement program, which produces some probability of detecting illegal fishing, and the penalty that will be applied to apprehended illegal fishers. This choice is the strategic variable for the leader. As the follower, the village decides in the second step, given the probability of detection and the expected penalty, how to allocate effort between legal and illegal fishing for the entire simulation time (50 yr). As such, the optimization is treated like a cooperative solution, in that the overall objective is to maximize the combined discounted net benefits of both fishing strategies. The simulation is run over 10 000 iterations and for a 50 yr time period. An artifact of models driven by profitability is that the players see the model's end (Year 50 in this model) as the 'end of the world,' and will therefore tend to catch as much as they can in the final years of the simulation. In the 'Results' section below, simulation outputs are thus discussed up to Year 45, with the final 5 yr of the simulations disre-

Table 1. Model biological and fishing parameters and sources; s : fishing strategy, where s represents legal fishing and $-s$ represents illegal fishing

| Variable | Symbol | Value | Source |
|--|--------|---|--|
| Snapper fishery | | | |
| Initial biomass (t) | B_0 | 6885 | Ainsworth et al. (2008) |
| Carrying capacity (t) | K | 16416 | Estimated from Ainsworth et al. (2008) |
| Maximum sustainable yield (t) | msy | 369 | Ainsworth et al. (2008) |
| Intrinsic rate of growth | r | 0.09 | Derived from Ainsworth et al. (2008) |
| Catch per trip (kg) | | $s = 5, -s = 8$ | Dohar & Anggraeni (2007), Pet-Soede et al. (1999) |
| Catchability | q | $s = 7.2 \times 10^{-7}, -s = 1.16 \times 10^{-6}$ | Derived from Dohar & Anggraeni (2007), Pet-Soede et al. (1999) |
| Unit price of fish (US \$ t^{-1}) | P | $s = 1260, -s = 1000$ | Dohar & Anggraeni (2007), Pet-Soede & Erdmann (1998) |
| Unit cost of effort (US \$ $trip^{-1}$) | co | $s = 3.25, -s = 3.00$ | Pet-Soede & Erdmann (1998), Pet-Soede et al. (1999) |
| Grouper fishery | | | |
| Initial biomass (t) | B_0 | 11565 | Ainsworth et al. (2008) |
| Carrying capacity (t) | K | 27702 | Estimated from Ainsworth et al. (2008) |
| Maximum sustainable yield (t) | msy | 1215 | Ainsworth et al. (2008) |
| Intrinsic rate of growth | r | 0.18 | Derived from Ainsworth et al. (2008) |
| Catch per trip (kg) | | $s = 11, -s = 5$ | Dohar & Anggraeni (2007), Kinch (2004) |
| Catchability | q | $s = 9.51 \times 10^{-7}, -s = 4.32 \times 10^{-7}$ | Derived from Dohar & Anggraeni (2007), Pet-Soede et al. (1999) |
| Unit price of fish (\$ t^{-1}) | P | $s = 4130, -s = 12800$ | Dohar & Anggraeni (2007), Ainsworth et al. (2008), Pet-Soede & Erdmann (1998) |
| Unit cost of effort (\$ $trip^{-1}$) | co | $s = 3.25, -s = 1.61$ | Pet-Soede & Erdmann (1998), Pet & Pet-Soede (1999) |

garded. The full solution algorithm to the optimization problem can be found in the Appendix.

Baseline simulations were run with no penalty costs of illegal fishing. In the optimal simulations, however, we assumed that the objective of the Regency government is to completely eliminate illegal fishing. There are many objective functions that could be imagined and for which simulations could be run. Our objective was to eliminate illegal fishing assuming that this (1) meets EBM goals; and (2) is precautionary. In this paper, we show the extent to which an enforcement program must go to eliminate incentives for illegal fishing. It is important to note that enforcement costs are not included in the net benefit calculation, i.e. any benefit derived from elimination of illegal fishing is gross of enforcement costs. We fully recognize the importance of measuring societal costs and benefits from enforcement in order to choose which conservation strategies deliver the highest benefits at the lowest cost, a process of conservation triage (Bottrill et al. 2008). However, the objective function, as is common in fisheries economics (see Clark 1990 for details), is based solely on resource rent derived from fishing. Because societal benefits, such as those emerging from EBM, are not included in the objective function, societal costs such as enforcement are not incorporated. To include enforcement costs would necessitate including benefits above and beyond fishing, which is outside the scope of this study, although within the realm of EBM (Tallis & Polasky 2009).

DATA

The following section outlines the data and assumptions used in the model (see Table 1). A subsection of the 'Results' presents a sensitivity analysis exploring how changes in some of the assumptions used affect the results of the model.

Snapper

The initial biomass (at $t = 1$) and carrying capacity (K) for the model were taken from the Raja Ampat Ecopath with Ecosim model (EwE) developed by Ainsworth et al. (2008). This model presented biomass estimates for 3 age classes of snapper, aggregated across 26 species: adult, sub-adult, and juvenile (see Ainsworth et al. 2008 for an explanation of species used in the EwE model). These estimates were added together to produce a biomass of 0.153 t km^{-2} (Ainsworth et al. 2008). The carrying capacity was estimated from the 1990 biomass estimates in the EwE report (Ainsworth et al. 2008).

We assumed that the 1990 biomass was about 20% lower than an unfished state, and multiplied the 1990 biomass estimates by 1.2 to estimate the carrying capacity, resulting in the use of $K = 16416$ t. The estimated initial and unfished biomasses were then multiplied by the study area, 45 000 km^2 , to give biomass estimates for all of Raja Ampat (Table 1).

The intrinsic rate of growth, r , was calculated using the equation:

$$r = \frac{4msy}{K} \quad (16)$$

where msy is the maximum sustainable yield (maximum catch) and K is the carrying capacity (Cadima 2003). The msy was taken from the Raja Ampat EwE model (Ainsworth et al. 2008). Catchability for snapper was calculated by dividing the average biomass of fish caught per trip by the total estimated biomass in the system. According to Dohar & Anggraeni (2007), the average artisanal fisher catches 5 kg of mixed snapper species per trip. This value was used for legal fishing in the model. Pet-Soede et al. (1999) reported that small-scale blast fishers catch about 8 kg of fish per trip, which is the value used here.

Price data used in the snapper model were taken from Dohar & Anggraeni (2007) and Pet-Soede et al. (1999). The average price of legal-caught adult snapper is about US \$1.26 kg⁻¹ (averaged over all legal gears; Dohar & Anggraeni 2007). Pet-Soede et al. (1999) estimated that for small-scale blast fishing, fishers received on average US \$1 kg⁻¹ for their catch. Pet-Soede et al. (1999) reported that the variable cost of small-scale blast fishing averaged US \$3.00 per trip. We assumed that blast fishing requires less time and thus requires less fuel than trips using legal gear. Therefore, 1 extra liter of diesel fuel was added to the legal cost of fishing (valued at US \$0.25 l⁻¹; Pet-Soede & Erdmann 1998), resulting in a cost per trip of US \$3.25 for legal gears.

Grouper

The same method described in the snapper section was used to estimate initial biomass ($t = 1$) and carrying capacity K for grouper. Ainsworth et al. (2008) reported an estimated 2006 grouper biomass, aggregated across the 3 age groups, of 0.257 t km⁻². The grouper carrying capacity was estimated by multiplying the 1990 EwE biomass estimate of 0.513 t km⁻² by 1.2, assuming that the unfished state is about 20% more than the 1990 biomass. The initial and unfished biomass estimates were then multiplied by the total marine area of Raja Ampat, 45 000 km², to determine initial biomass and carrying capacity. Again, the intrinsic rate of growth, r , was calculated as shown for snapper. The msy parameter was taken from the Raja Ampat EwE model (Ainsworth et al. 2008). The catchability coefficients used in the grouper model were calculated in the same manner as for snapper.

Dohar & Anggraeni (2007) reported that the average artisanal fisher catches about 11 kg of mixed grouper per trip. Pet & Pet-Soede (1999) reported that small-scale cyanide operations catch 1 kg of fish per trip, and medium-scale operations catch up to 20 kg. We assumed in this model that the average catch per trip for small-scale cyanide fishers is 5 kg. These 2 production values, 11 kg and 5 kg, were divided by the total grouper biomass to give the catchability coefficients used in the model.

The average price of legal-caught grouper in Raja Ampat is about US \$5.60 kg⁻¹ (averaged over all legal gear types), according to Dohar & Anggraeni (2007). Ainsworth et al. (2008), however, used an average price of US \$2.64, which includes adult and sub-adult grouper. For the model, the average of these 2 estimates was used: US \$4.13 kg⁻¹. A price of US \$7.50 kg⁻¹ was used in the EwE model for the average unit price of cyanide-caught grouper (Ainsworth et al. 2008). However, Pet-Soede & Erdmann (1998) suggested that fishers can receive upwards of US \$18.80 kg⁻¹ for live coral trout. Here, we used the average of these 2 estimates, US \$12.80 kg⁻¹, in the model. The unit cost of US \$3.25 per trip for legal fishing estimated in the case of snapper was also used in the grouper model, as the same (legal) gear is used to target both types of fish. Pet & Pet-Soede (1999) reported that cyanide is quite cheap, with a small-scale cyanide operation using about 1 l of cyanide per trip, at a cost of \$1.11. We therefore took the cost estimate of Pet-Soede & Erdmann (1998) for blast fishing, subtracted the cost of the locally-made bombs (\$2.50 per trip), and added in the cost of cyanide (\$1.11), resulting in a cost estimate of US \$1.61 per trip.

Table 1 outlines the model parameter inputs. A sensitivity analysis was conducted to test how assumptions on input parameters affected the results. The key variables tested in the sensitivity analysis were the externalities term (α), the discount rate, the carrying capacity of the system, and the price. The results of this analysis are presented following the results of the main simulations.

RESULTS

Snapper fishery

Baseline

With no formal monitoring and enforcement program in Raja Ampat to detect and punish fishers

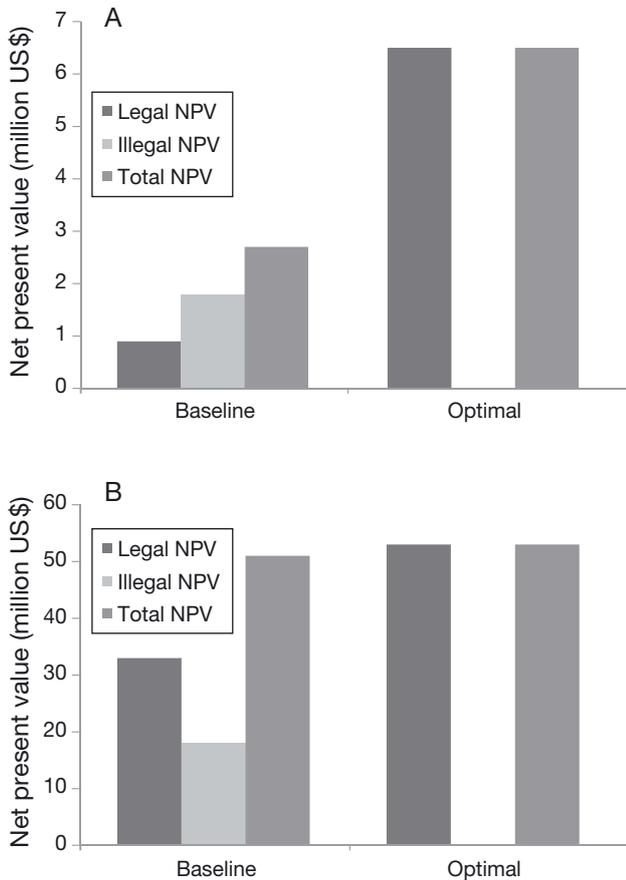


Fig. 1. Comparison of legal, illegal, and total net present value (NPV) between baseline and optimal simulations for (A) snapper and (B) grouper

using destructive gears (M. Erdmann pers. comm.), the first model simulation was run such that there are no extra private costs associated with illegal fishing (i.e. $\rho \times Fee \times E_{-s,t} = 0$). This is the 'baseline' scenario against which the 'optimal' simulation is compared. In the baseline, the total discounted net present value (NPV, the discounted net benefits summed over the simulation time of 45 yr) from legal fishing is about US \$0.9 million. The NPV from blast fishing is almost twice this, at US \$1.8 million. The total NPV from both types of fishing is the summation of these two, and is equal to almost US \$2.7 million (Fig. 1A).

It is currently more profitable to fish snapper using bombs, rather than legal methods, as indicated by the effort trends (Fig. 2A). For the most part, over all time periods more effort is allocated to blast fishing than to legal fishing. The higher effort level, along with the assumed higher catchability of blast fishing, leads to a greater catch by blast fishing at all time periods (Fig. 2B). Although catch initially increases through time (due to increased effort and an initial biomass increase), the future decrease in biomass leads to declining catches near the end of the model. Over the 45 yr period, about 10 000 t of snapper are caught in total, with an annual average catch of about 220 t. In the baseline, there is a decrease in snapper stock biomass over time to about 4000 t (Fig. 2C), and the net benefits from blast fishing are greater than those for legal-caught methods in all time periods (Fig. 2D).

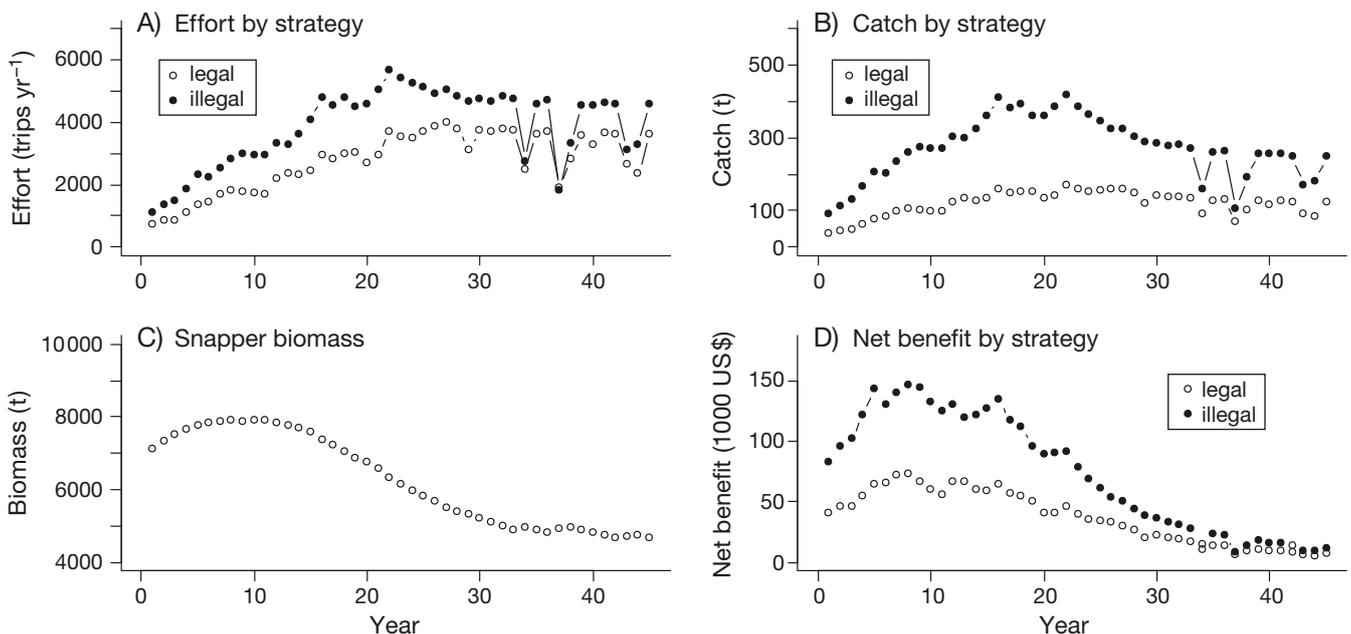


Fig. 2. Snapper baseline simulation of (A) effort, (B) catch, (C) biomass, and (D) net benefit

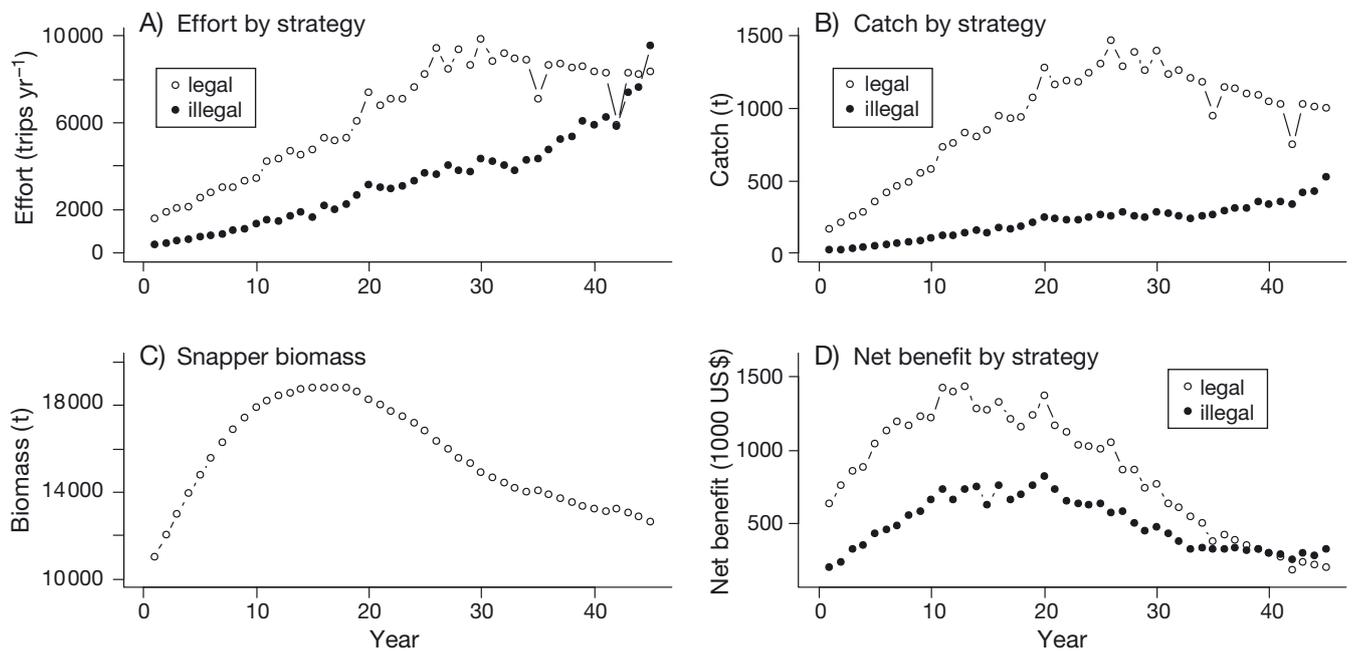


Fig. 3. Grouper baseline simulation of (A) effort, (B) catch, (C) biomass, and (D) net benefit

Optimal

In line with EBM, thus assuming that the objective of the government is to totally eliminate blast fishing, the simulations were re-run at increasing probabilities of detection and penalty fees (see Fig. 4). In completely eliminating blast fishing, the NPV of the artisanal snapper fishery increases from US \$2.7 million to US \$6.5 million over the 45 yr period (Fig. 1). Over the 45 yr period, a total of about 18 000 t of snapper are caught, all with legal methods, averaging 420 t yr⁻¹. In Year 45 of the simulation, biomass is stabilized at around 9000 t. This is just over twice the biomass in Year 45 of the baseline.

Grouper fishery

Baseline

The baseline scenario is one which assumes that the status quo of zero monitoring and enforcement continues in Raja Ampat for the next 45 yr. Under this scenario, the fishery yields US \$51 million in total NPV over the 45 yr (Fig. 1B). Fig. 3A,B shows the effort and catch profiles for the baseline solution. Over 45 yr, just over 44 000 t of grouper are caught in total, averaging about 9300 t annually. More effort is allocated to, and more catch is taken by, legal

methods in all years, although effort converges near Year 45. The price of illegally caught grouper is higher, and the cost lower, and but the high catchability of legal grouper fishing means fishers are spending more effort fishing with legal gears. The decrease in legal effort, and increase in illegal effort, at the end of the simulation, is probably driven by the expected 'end of the world' behavior described earlier. Grouper biomass increases at the start of the simulation, but after reaching its maximum at about Year 20, biomass starts to decrease for the remaining time steps (Fig. 3C), stabilizing around 12 000 t.

Optimal

The optimal solution assumes that the government is trying to completely eliminate cyanide fishing with a combined probability of detection and fine (Fig. 4). With the elimination of the cyanide fishery for grouper, the total NPV over the 45 yr is US \$53 million (Fig. 1B). With the total elimination of the illegal fishery, the value of the grouper fishery is worth about US \$2 million more over the 45 yr, which is an increase of only about 5%. Over the 45 yr, a total of almost 56 000 t of grouper are removed from the ecosystem, i.e. ~1240 t yr⁻¹. In the optimal simulation, the biomass stabilizes at a level around 10 500 t, which is 1000 t higher than in the baseline.

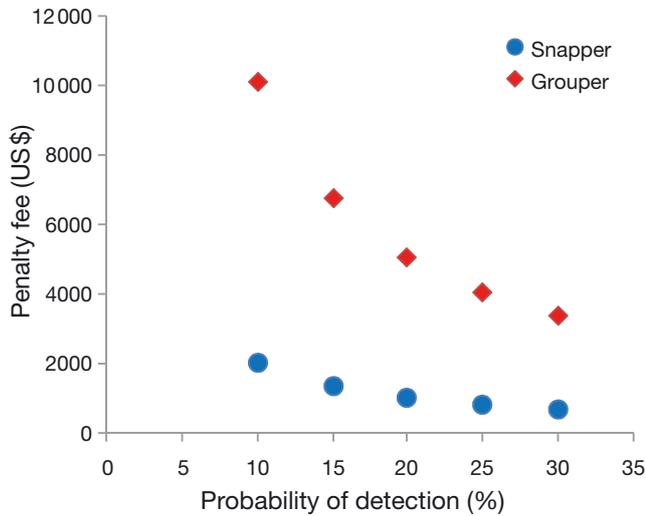


Fig. 4. Combinations of probability of detection and penalty fee required to eliminate dynamite fishing for snapper and cyanide fishing for grouper

Enforcement

Both of the Regency enforcement decision variables, penalty and fee, need to take on values that are both positive and realistic. Mathematically, they are multiplied with effort in the cost function, so if either value is 0, the fisher faces no extra cost from illegal fishing. Several combinations of detection probabilities and fees are possible to reach the desirable solution of no blast fishing. The Raja Ampat Regency government would have to evaluate the possible combinations to determine which meet their budget and fisheries management plans. There is a direct trade-off between investing a lot in detecting power (monitoring), versus investing little but imposing a higher fine when fishers are apprehended, and possibly unrealistic amounts (enforcement) (Fig. 4).

Table 2. Discount rate sensitivity analysis for grouper. NPV: net present value (in million US \$)

| Discount rate | 0 ^a | 4 | 7 | 10 |
|------------------------------|----------------|-------|-------|-------|
| Legal catch (t) | 31400 | 29400 | 36400 | 33600 |
| Illegal catch (t) | 3660 | 4900 | 7710 | 11200 |
| Total catch (t) | 35010 | 34300 | 44100 | 44800 |
| Illegal catch proportion (%) | 10 | 14 | 17 | 25 |
| Legal NPV | 126 | 50 | 33 | 27 |
| Illegal NPV | 45 | 24 | 18 | 18 |
| Total NPV | 171 | 74 | 51 | 45 |
| Biomass (t), Year 45 | 21000 | 19900 | 12000 | 9300 |

^aThis is actually $\lim_{t \rightarrow \infty} B_t$ to allow convergence; meaning it is the limit as it approaches zero but never actually reaches zero (Clark 2006)

This statement assumes that it would cost the government more to increase their probability of detection from 10%, to 15%, to 20%, etc. For each of the detection probabilities tested (5% to 30%), a penalty fee 5 times higher is required to eliminate cyanide fishing compared with blast fishing (Fig. 4).

We tested how the assumptions in some of our input parameters changed the simulation outcomes, including assumptions about the discount rate and intrinsic growth rate (for the grouper fisher) and the carrying capacity and price (for the snapper fishery), and the EBM α parameter.

Discount rate

Sustainability implies that the present generation's use of a resource does not prevent future generations from enjoying the same resource. It is known that high rates of discount tend to result in societies over-exploiting their resources today (Clark 1990). The simulation results presented for the snapper model above were created by assuming a 7% discount rate (a discount factor of 0.935). To examine the effect of the discount rate on baseline catch, biomass, and economic value of the fishery, simulations were re-run with varying discount rates, with the results for the grouper fishery shown in Table 2. As predicted by the literature (Clark 1973, 1990, Sumaila 2004, Berman & Sumaila 2006), higher rates of discount lead to a lower stock size in the future, as well as a lower NPV of the fishery. From an EBM policy perspective, it is particularly interesting to examine how the optimal fee varies with the discount rate. When the optimal simulations are run with a higher discount rate (10%), and keeping the detection probability the same, a higher fee is required to eliminate illegal fishing in each case, given the higher discount rate. This is an important issue for the Regency government to consider when instituting an incentive scheme, given that researchers have argued that small-scale reef fishers have discount rates high in excess of those usually used in bioeconomic modeling (Teh et al. 2015).

Intrinsic rate of growth

The intrinsic rate of growth, r , is a biological parameter defining how quickly a population reproduces (considering natural mortality). Table 3 shows the results of the sensitivity analysis for the grouper simulations with the low and high values used in the

simulation given on either side of the main value used in the model. A higher r value implies a more productive population, thus providing for larger catches and value through the 45 yr (Table 3). Interestingly, however, regardless of the intrinsic growth rate assumption, the NPV of baseline and optimal solutions remains almost negligible in difference.

Table 3. Intrinsic growth rate sensitivity analysis for grouper. NPV: net present value (in million US \$)

| Growth rate | | Low (0.09) | Model (0.18) | High (0.36) |
|----------------------|----------|---------------|-----------------|----------------|
| Biomass (t), Year 45 | Baseline | 11700 | 11800 | 16260 |
| | Optimal | 8728 | 10500 | 14840 |
| Total catch (t) | Baseline | 23700 | 43800 | 73000 |
| | Optimal | 30300 | 55600 | 94000 |
| NPV | Baseline | 27 | 51 | 91 |
| | Optimal | 26 | 53 | 90 |

Table 4. Carrying capacity sensitivity analysis for snapper. NPV: net present value (in million US \$)

| Carrying capacity (t) | | Low (13680) | Model (16416) | High (20520) |
|-----------------------|----------|----------------|------------------|-----------------|
| Biomass (t), Year 45 | Baseline | 4400 | 4400 | 4400 |
| | Optimal | 7400 | 8900 | 10200 |
| Total catch (t) | Baseline | 9200 | 10600 | 12000 |
| | Optimal | 15100 | 17900 | 24400 |
| NPV | Baseline | 2.2 | 2.7 | 3.2 |
| | Optimal | 5.1 | 6.2 | 8.6 |

Table 5. Price sensitivity analysis for snapper. NPV: net present value (in million US \$)

| Legal price (US \$ t ⁻¹) | | Low (945) | Model (1260) | High (1575) |
|--|----------|--------------|-----------------|----------------|
| Biomass (t), Year 45 | Baseline | 4300 | 4200 | 4440 |
| | Optimal | 8400 | 8900 | 8900 |
| Total catch (t) | Baseline | 10400 | 10600 | 9900 |
| | Optimal | 18800 | 17920 | 18100 |
| NPV | Baseline | 2.5 | 2.7 | 3.0 |
| | Optimal | 5.0 | 6.2 | 8.0 |
| Illegal price (US \$ t ⁻¹) | | Low (750) | Model (1000) | High (1250) |
| Biomass (t), Year 45 | Baseline | 4300 | 4200 | 4400 |
| | Optimal | 9500 | 8900 | 9500 |
| Total catch (t) | Baseline | 10600 | 10603 | 10500 |
| | Optimal | 17900 | 17920 | 17900 |
| NPV | Baseline | 2.3 | 2.7 | 3.0 |
| | Optimal | 6.2 | 6.2 | 6.2 |

Carrying capacity

The carrying capacity (K) used to run the model was calculated by multiplying the 1990 snapper biomass by 1.2. The model was re-run using a lower K estimate, by assuming that the 1990 biomass was the un-fished state, and a higher K , assuming that K is actually 1.5 times the 1990 biomass. As would be expected, a larger snapper carrying capacity leads to a greater catch and a higher NPV over time. Obviously, if the model simulations were being used to recommend allowable catches, it would be important to understand and quantify the uncertainty around this parameter, and thus in the catch estimates. Table 4 presents the biomass in Year 45, total catches, and NPV over the 45 yr period at varying K values. The low and high values used in the sensitivity analysis simulation are given on either side of the main value used in the model. The relative profitability of the optimal solution (versus the baseline) ranges from about 2 to 2.7, that is, a higher carrying capacity leads to a higher relative profit of the optimal solution.

Price

The price of legally caught snapper used in this model is US \$1260 t⁻¹. The bounds for the sensitivity analysis were calculated by multiplying this price by 0.75 to get the low price bound and 1.25 to get a high price bound. The model was then rerun with these bounds (Table 5). A similar method was used to calculate the low and high bounds for illegally caught snapper: the base price of US \$1000 t⁻¹ was multiplied by 0.75 and 1.25. Changes in price do not tend to change the biomass at the end of the simulation time, nor the total amount of catch over the 45 yr, but they do obviously change the value of the catch. As would be expected, the relative profitability of eliminating illegal fishing increases with higher prices for legally caught fish. Conversely, when illegally caught snapper fetches a higher price, the relative profitability of eliminating that fishing strategy decreases. The low and high values used in the sensitivity analysis simulation are given on either side of the main value used in the model.

Externalities and EBM

By including an externalities parameter, namely α , we can test whether changes in behavior would

result from incorporating EBM thinking into decision making. In the proceeding results, we assumed ecosystem impacts of illegal fishing above and beyond illegal fishing by utilizing the α parameter and assuming it equal to 2. This assumption was in keeping with EBM. We tested the value of α for both snapper and grouper simulations (Fig. 5). Higher values of α force fishers to fish less due to ecosystem signals they receive from illegal fishing. With $\alpha > 1$, each unit of illegal fishing essentially means more removals but no ex-vessel price associated with that removal. When ecosystem impacts are included, less illegal fishing is chosen by villages. Because of this ecosystem signal, and therefore less illegal fishing compared to no ecosystem signal, the calculated probability of detection and fine combinations are less here than they otherwise would be. The Regency government needs to consider this when setting policies. Higher enforcement may be necessary when ecosystem signals are weak, which is often the case with fisheries externalities.

DISCUSSION

The use of destructive fishing gears threatens fisheries, marine biodiversity, and ecosystem services worldwide (Pauly 1989, Pet-Soede & Erdmann 1998, Cesar et al. 2000, Halim & Mous 2006). In Raja Ampat, with artisanal fisheries currently valued at US \$7 million (Dohar & Anggraeni 2007), it seems evident that ensuring sustainable fishery yields through time should be a priority for the government. Although sustainable fisheries management requires several components, the elimination of illegal fishing is certainly an important one (FAO 2001, 2007). The perverse incentives to fish using explosives and cyanide are demonstrated in this analysis. Effort is allocated to these fishing methods due to their profitability as apparent in the baseline simulations, confirming anecdotal reports that the use of these destructive fishing gears is common in Raja Ampat. The current analysis suggests that if the present-day situation continues, with no monitoring and enforcement by the government, the use of explosives and cyanide in Raja Ampat may result in lower catches and less effort in the fisheries through time. As the government wishes to use the fisheries sector to increase the standard of living for Regency citizens (M. Wanma pers. comm.), sustainability of the artisanal sector is vital.

In this analysis, the artisanal snapper fishery is estimated to be worth between US \$2.7 and \$6.5 million

over the next 45 yr. The elimination of explosives on the reef could result in a higher stock biomass, and fairly consistent catches through time. It appears that the optimal solution is perhaps a desirable one for the Regency government from an EBM perspective as eliminating illegal fishing leads to increased benefits. The recent rise in tourism (E. Frommenwiler pers. comm.) and pearl farming (M. Erdmann pers. comm.) in Raja Ampat has resulted in a perceived decrease in the number of blasts occurring in the area. The presence of dive operations out on the water, as well as armed guards present at the farms, could potentially act as pseudo enforcers, perhaps decreasing the government's management costs. However, while eliminating blast fishing brings economic benefits, the potential benefits from eliminating cyanide fishing are not so clear. Cyanide fishing tends to target grouper spawning aggregation sites, thus possibly leading to recruitment overfishing (Cesar et al. 2000). Researchers in Raja Ampat have suggested that the amount of cyanide fishing has been decreasing in the area, but evidence suggests that the price of live-caught grouper is still high. This high price, coupled with the current inability of managers to charge cyanide fishers with a crime, may be continuing to incentivize illegal fishing, even though grouper populations have declined in Indonesia and may be harder to target (Halim 2003).

Several possible combinations of detection probabilities and fisher fines were presented. Although it is not our intention to suggest which combination is best, it is important to note that the potential for bribes in developing countries is often large (Owino 1999, De Lopez 2003). As such, it might be in the government's best interest to invest heavily in monitoring, meaning a higher detection probability and lower fines. The high profitability of the cyanide fishery means that low detection probabilities would require unfathomable fines. Charles et al. (1999) recommended that the penalty fee does not exceed the assets of the apprehended fisher, and other research suggests that probability of detection may in fact be more important than the penalty in dissuading non-compliance (Furlong 1991). These insights, coupled with the reality in Papua Province (where the per capita gross domestic regional product is less than US \$1000; Bappeda 2004 in Dohar & Anggraeni 2007), lead to the conclusion that fining fishers an exorbitant amount and expecting payment may be unrealistic.

It has been argued that the potential benefits of decreasing illegal fishing should be assessed to help determine the optimal cost investment into an

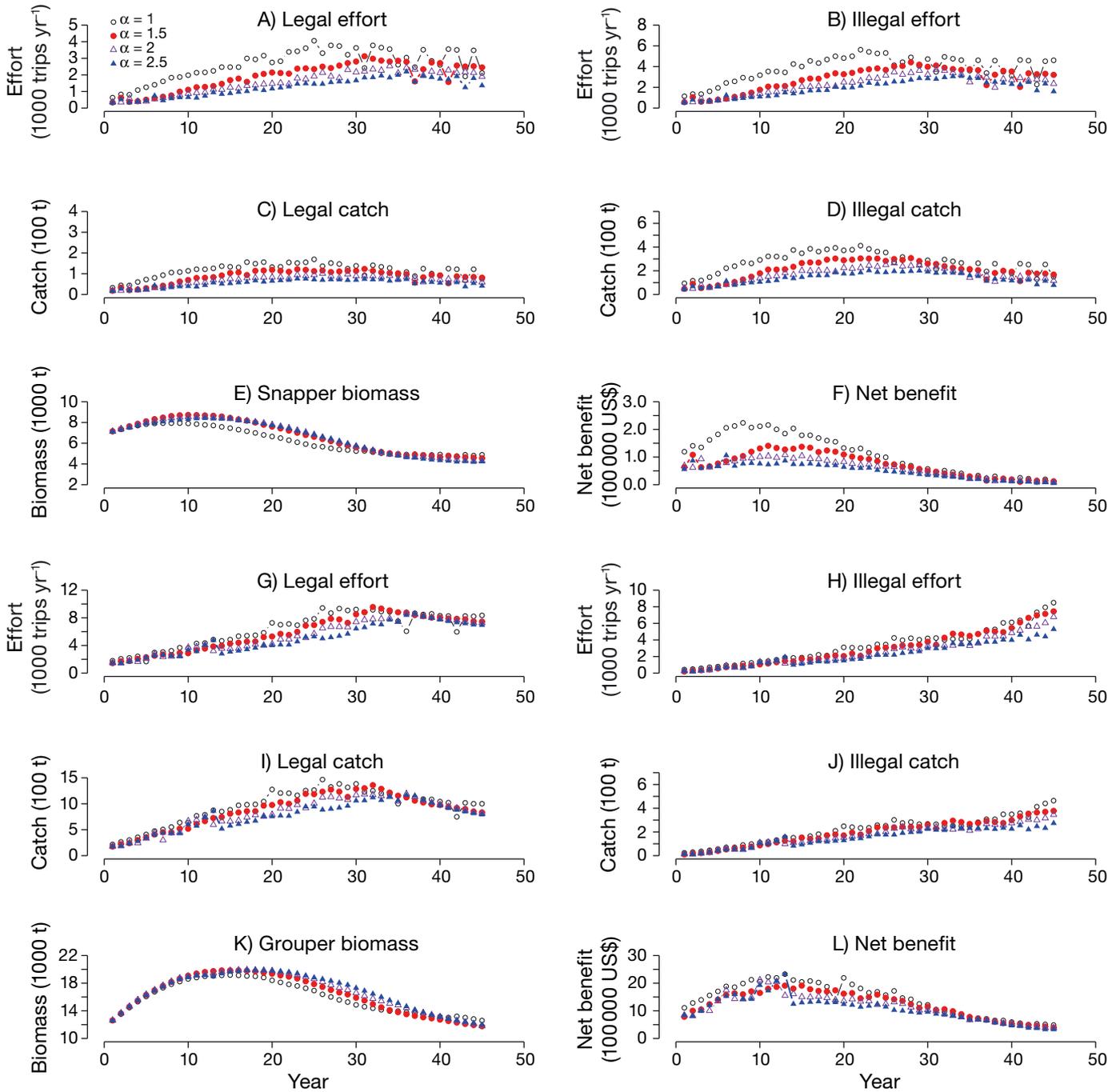


Fig. 5. (A–F) Snapper and (G–L) grouper simulations run with increasing levels of the ecosystem parameter α

enforcement initiative (Furlong 1991). In this analysis, we have quantified the potential benefits arising from implementation of an enforcement regime, but have not tried to determine enforcement optimality given the costs required to implement such a regime. While the leader is the Regency government, it could be that other parties contribute to monitoring and en-

forcement in the region—for example, the federal government (Ministry of Marine Affairs and Fisheries) or non-governmental conservation organizations. Who should bear these costs is a valid question; one that has been raised before (Balmford & Whitten 2003, Rangeley & Davies 2012). The debate over the costs has to take place within the EBM framework,

which would mean outside the realm of the Indonesian government. Who seeks to benefit from coral reef conservation? Arguably, those who will likely benefit should be part of the cost assessment.

In addition to the mere presence of a monitoring and enforcement program in Raja Ampat, initiation of some type of communication platform is equally important. This process, known as 'sosialisasi' or socialization in Indonesian, is something like education, but is really aimed at providing legitimacy to given initiatives. Individuals, and quite possibly society as a whole, are more likely to follow the rules when they are viewed as being legitimate (Tyler 1990). In addition, there is evidence that the decision to not fish illegally may be influenced by things other than economics disincentives, for example moral and social considerations (Kuperan & Sutinen 1998, Sutinen & Kuperan 1999), both of which are important in customary marine tenure. For these reasons, our approach in this paper was not to seek the most economically efficient means of eliminating illegal fishing, but rather to offer insights into the benefits that could accrue from enforcement (in the absence or presence of EBM—operationalized by the α parameter), and raise some points of consideration, for example an understanding of fisher discount rates.

CONCLUSION

Munro (1992) explained, in general economic terms, that the present-day investment in a stock of capital will benefit a society by increasing the society's productive capacity in the future. By eliminating destructive fishing methods, the Raja Ampat Regency could be ensuring a flow of benefits to the community through time. Furthermore, other commercially targeted fish, such as trevally and fusiliers, as well as the prized Napoleon wrasse, would most likely benefit from reduced destructive fishing methods. What is also important to consider is that destructive fishing not only jeopardizes fish stocks, but also the very ecosystems that commercial species depend on (Pauly 1989, Cesar et al. 2000). This analysis incorporated the negative impacts of fishing on just the target stock; however, reefs exposed to blast fishing show reduced diversity. Furthermore, modeling simulations run by Ainsworth et al. (2008) linked destructive fishing to available coral habitat for refuge spaces, and illustrated that coral destruction affects dependent fish populations both acutely and chronically, due to the low regeneration time for corals.

One of the more difficult pills to swallow with any EBM plan is that short-term costs are usually necessary in order to attain longer-term benefits (Sumaila 2005). Who should bear these costs is an important debate unfolding in the conservation literature. With many dynamic partnerships currently evolving in coral reef conservation, hopefully the debate will be fruitful. In any event, EBM explicitly recognizes the impacts that fishing has on the ecosystem (Ward et al. 2002), and in effect forces decision makers to compare possible management plans to a more realistic present-day status quo, one that includes EBM principles. From the results of this analysis, we conclude that the Raja Ampat Regency government may benefit by incorporating the ecological externalities of destructive fishing into their EBM planning. Staying with the true nature of EBM, villages in Raja Ampat, which Regency citizens identify as being responsible for marine management, need to be included in fisheries sector planning and educated on the destructive nature of, and lost revenue due to, the frequent use of destructive fishing gears.

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Appendix

Lagrangian function

A modified Lagrangian function is used in this model to solve for the maximization problem facing fishers, subject to the constraints of the model. The natural biological constraint that $B_t \geq 0$ must be met, and thus the model applies a penalty (γ_t) when $B_t < 0$:

$$L_t(B_t, E_{s,t}, \gamma_t) = \delta^t \pi + \gamma_t \varphi^-(B_t, E_{s,t}), \quad \forall s \quad (\text{A1})$$

where the term φ represents the constraint function for which the modified Lagrange multiplier, γ_t , is applied only in the case when $\varphi < 0$. That is, φ is given by $\min(0, \varphi)$ (Flåm 1993). The profit and constraint functions are expanded in the following equation to give the entire Lagrangian:

$$\begin{aligned} L(B_t, E_{s,t}, E_{-s,t}, \gamma_t) = & \delta^t \sum_{t=0}^T \left(q_s E_{s,t} B_t P_s - \frac{c o_s E_{s,t}^{1+b}}{1+b} \right) \quad (\text{A2}) \\ & + \delta^t \sum_{t=0}^T \left(q_{-s} E_{-s,t} B_t P_{-s} - \frac{c o_{-s} E_{-s,t}^{1+b}}{1+b} - \rho Fee E_{-s,t} \right) \\ & + \gamma^t \left[r B_t \left(1 - \frac{B_t}{K} \right) - (q_s E_{s,t} B_t) - \alpha (q_{-s} E_{-s,t} B_t) \right] \end{aligned}$$

where α is used to model the externality of blast and cyanide fishing on reef habitat.

Solution algorithm

The solution algorithm used in this analysis is modeled after Flåm (1993) and Sumaila (1995), assuming a cooperative outcome. The partial differentials for the effort, biomass, and multiplier adjustments are derived in this sec-

tion in order to identify the rates of change of effort, biomass, and the multiplier. For these equations, a switch function is used, and denoted $H(r)$. Let $H(\&) = 1$ when $\& < 0$, and $H(\&) = 0$ otherwise. Thus, $H(\&)$ attains a value of 1 when a constraint is violated.

Effort adjustment: How does the Lagrangian function change with respect to a change in effort? This is in fact the agent's decision variable of the model.

First, we consider the adjustment of legal effort:

$$\frac{\partial L_t}{\partial E_{s,t}} = \delta^t (q_s B_t P_s - c o_s E_{s,t}^b) + \gamma^t [H(\varphi)(-q_s B_t)] \quad (\text{A3})$$

By expanding the function, we have:

$$\begin{aligned} \frac{\partial L_t}{\partial E_{s,t}} = & \delta^t (q_s B_t P_s - c o_s E_{s,t}^b) \quad (\text{A4}) \\ & + \gamma^t \left[H \left(r B_t \left(1 - \frac{B_t}{K} \right) - q_s E_{s,t} B_t - \alpha q_{-s} E_{-s,t} B_t \right) (-q_s B_t) \right] \end{aligned}$$

Now the adjustment of illegal effort, $-s$:

$$\frac{\partial L_t}{\partial E_{-s,t}} = \delta^t (q_{-s} B_t P_{-s} - c o_{-s} E_{-s,t}^b - \rho Fee) + \gamma^t [H(\varphi)(-\alpha q_{-s} B_t)] \quad (\text{A5})$$

Biomass adjustment: How does the Lagrangian function change with respect to a change in the biomass? Here, we consider the first order partial differential with respect to biomass:

$$\frac{\partial L_t}{\partial B_t} = \delta^t (q_s E_{s,t} P_s + q_{-s} E_{-s,t} P_{-s}) + \gamma^t \quad (\text{A6})$$

$$\left[H(\varphi) \left(-1 + r - \frac{2r B_t}{K} - q_s E_{s,t} - \alpha q_{-s} E_{-s,t} \right) \right]$$

Multiplier adjustment: How does the Lagrangian function change with respect to a change in the multiplier?

$$\frac{\partial L_t}{\partial \gamma_t} = -H(\varphi) \left(r B_t \left(1 - \frac{B_t}{K} \right) - q_s E_{s,t} B_t - \alpha q_{-s} E_{-s,t} B_t \right) \quad (\text{A7})$$